Evaluation of restoration and management actions in the Molopo savanna of South Africa: an integrative perspective

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It all starts here ™

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Abstract

The loss of ecosystem resilience and rangeland (often referred to as veld in South Africa) productivity is a major problem in the semi-arid Savanna environments of southern Africa. The over-utilization of rangelands in the Molopo region of the North-West Province in South Africa has resulted in profound habitat transformations. A common regional indicator of rangeland degradation is the imbalance in the grass-woody ratio, characterized by a loss of grass cover and density with increased shrub or tree density. This can result in major reductions of rangeland productivity for the grazing animal, forcing land users to apply active or passive restoration actions to improve rangeland condition, control the thickening of woody species (bush thickening), mitigate economic losses and restoring the aesthetical value of the Savanna environment for ecotourism and game hunting aspects.

This study formed part of the multinational EU-funded PRACTICE project ("Prevention and restoration actions to combat desertification: an integrated assessment"). The first aim of the study was to evaluate locally applied restoration actions using a participatory approach, followed by interviews with certain stakeholders that formed part of a multi-stakeholder platform (MSP) related to the livestock and game farming community in the Molopo. Participants of the MSP ranked indicators according to their relative importance regarding the restoration actions on an individual basis. The individual ranking results were combined with quantitative bio-physical and qualitative socio-economic measurements for each indicator in a multi-criteria decision analysis (MCDA), whereby the alternative actions were ranked according to their relevancy and performance. The results were then shared with members of the MSP in order to stimulate discussion among the members and contribute to the social learning of the project outcome.

The overall positive response and acceptance of results by members of the MSP changed the perceptions and objectives of the land users regarding rangeland management. This type of participatory assessment was therefore found to be very promising in helping to identify more sustainable actions to mitigate rangeland degradation in the Molopo Savanna region. There is, however, still an urgent need to create legal policy frameworks and institution-building, to support local-level

implementation in all socio-ecological and economic settings, particularly in communal areas.

The second aim was to evaluate the effect of two chemical bush control actions (chemical hand- (HC) and aeroplane control (AC)) as well as rotational grazing (RGM) on the Molopo Savanna vegetation.

Results show that rangeland productivity, i.e. forage production and grazing capacity, was found to be negatively related to the woody phytomass in the savanna system studied. Bush thickening influenced grass species composition which was commonly associated with a decline in the abundance of sub-climax to climax grasses, respectively. All three actions (HC, AC & RGM) significantly reduced the woody phytomass and increased forage production and grazing capacity.

Although AC resulted in the highest reduction of woody phytomass, the highest forage production and grazing capacity was found under RGM. The second highest grazing capacity was found in HC sites, which was due to a high abundance of perennial, palatable climax grass species. Results from this study also show that the patterns and compositions of grass species, grass functional groups (GFGs) and woody densities indicated by RGM and chemical HC, best resemble a productive and stable savanna system that provides important key resources to support both grazing and browsing herbivores.

Keywords: Integrated assessment; stakeholder participation; indicator identification; bush thickening; chemical control; rangeland condition; grass:woody ratio.

Opsomming

Tans is die verlies aan weidingsproduktiwiteit 'n groot probleem in die semi-droë Savannas van suidelike-Afrika. Die oorbenutting van die weivelde in die Molopo streek van die Noordwes-provinsie in Suid-Afrika het gelei tot ekstreme habitat transformasies. 'n Algemene aanduiding van landdegradasie is die wanbalans in die gras en houtagtige verhouding wat gekenmerk word deur 'n verlies van grasbedekking met 'n toename in die digthede van struike of bome. Dit kan lei tot afnames in die weidingsproduktiwiteit wat landgebruikers dwing om aktiewe of passiewe restourasiestappe te neem om hierdeur die weidingskapasiteit te verbeter, die toename in houtagtige plante (bosverdigting) te beheer om sodoende ekonomiese verliese te voorkom en die Savanna habitatte te herstel vir eko-toerisme en wildjag doeleindes.

Die studie het deel gevorm van die multinasionale PRACTICE projek (“Prevention and restoration actions to combat desertification: an integrated assessment”) wat deur die Europese Unie gefinansier is. Die eerste doel van die studie was om restourasiepraktyke wat plaaslik geïmplementeer word te evalueer deur 'n deelnemende benadering te volg met onderhoude wat gevoer is met ’n groep plaaslike belanghebbendes van die Molopo vee- en wildboerderygemeenskap. Deelnemers in die studie het indikatore individueel gerangskik volgens relatiewe belangrikheidwaardes in verband met restourasiepraktyke. Die resultate van die individuele rangskikking is daarna gekombineer met data wat verkry is van kwantitatiewe biofisiese en kwalitatiewe sosio-ekonomiese opnames vir elke indikator in 'n multi-kriteria besluitnemings analise, waardeur die alternatiewe praktyke gerangskik was volgens elkeen se toepaslikheid en prestasie. Terugvoer op die resultate was gelewer aan die groep belanghebbendes om bespreking onder mekaar te bevorder wat bygedra het tot een van die projek uitkomste naamlik sosiale leer.

Die algehele positiewe terugvoer en ook aanvaarding van die resultate deur die groep belanghebbendes het gelei tot die verandering in die persepsies van die landgebruikers in verband met weidingsbestuur. Hierdeur is gevind dat die soort deelnemende evaluering baie belowend is om meer volhoubare praktyke te

identifiseer om sodoende weidingagteruitgang te voorkom in die Savanna streek van die Molopo. Daar is egter steeds 'n dringende behoefte aan wetlike beleidsraamwerke om die implementering daarvan op ’n plaaslike vlak in alle sosio-ekologiese en ekonomiese omgewings te ondersteun, veral in die kommunale gebiede.

Die tweede doel van die studie was om die effek van twee chemiese beheeraksies van houtagtige plante (chemiese hand- (HB) en vliegtuig beheer (VB)) sowel as wisselweiding (WW) in die Molopo Savanna te evalueer.

Daar is bevind dat die weidingsproduktiwiteit, d.w.s. die voerproduksie en weidingskapasiteit, negatief verband hou met die digthede van houtagtige plante in die Savanna sisteem. Die verdigting van houtagtige plante beïnvloed die gras spesiesamestelling negatief wat veral geassosieer was met 'n afname in sub - klimaks en klimaks grasse. Al drie die bogenoemde aksies (HB, VB en WW) het gelei tot ’n aansienlike afname in die digthede van houtagtige plante en daarmee gepaard ’n toename in die voerproduksie en weidingskapasiteit van die weivelde.

Hoewel VB gelei het tot die hoogste afname in die digthede van houtagtige plante, was daar gevind dat die hoogste voerproduksie en weidingskapasiteit gevind is binne WW. Die tweede hoogste weidingskapasiteit was gevind binne HB, wat te wyte was aan die volopheid van meerjarige, smaaklike klimaksgrasspesies. Die gras spesiessamestellings, gras funksionele groepe en die digthede van houtagtige plante van WW en chemiese HB toon 'n meer gebalanseerde savanna sisteem wat belangrike sleutel hulpbronne vir beide gras- en blaar vretende herbivore in hou.

Sleutel-woorde: Geïntegreerde assessering; deelname van belanghebbendes; indikator identifikasie; bosverdigting, chemiese beheer, gras-houtagtige verhouding.

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All the flowers of all the tomorrows are in the seeds of today - Indian Proverb.

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Glossary of Abbreviations

AC - Aeroplane control

ANOSIM - Analysis of similarity

BE - Bush encroached

BT - Bush thickened

BTS - Bush thickened sites

BU - Browser unit

CC - Chemical control

DARD - Department of Agriculture and Rural Development

DM - Dry matter

DPM - Disc Pasture Meter

EC - Electrical Conductivity

ETTE - Evapotranspiration Tree Equivalents

FIXMOVE - Fixed point monitoring of vegetation methodology

HC - Hand control

IAPro - Integrated Assessment Protocol

KCl - Potassium chloride

LFA - Landscape function analysis

LRAD - Land Redistribution for Agriculture Development

LSU - Large Stock Unit

MEA - Millennium ecosystem assessment

MAP - Mean annual precipitation

MSP - Multi-stakeholder platform

NRG - No rotational grazing

NW-DARD - North-West Province's Department of Agricultural and Rural Development

NWU - North-West University

PAR - Participatory action research

PCA - Principal component analysis

PCQ - Point centre quarter

PRACTICE - Prevention and restoration actions to combat desertification: an integrated approach

r_s - Spearman's rank correlation coefficient

RGM - Rotational grazing management

ROAM - Return On Asset Managed

ROI - Return on Investment

RV - Re-vegetation

SHs - Stakeholders

SIMPER - Similarity percentage

SOC - Soil organic carbon

TE - Tree equivalent

UNCCD - United Nations Convention to Combat Desertification

UNEP - United Nations Environment Program

WMO - World Meteorological Organization

1

Introduction

1.1 General introduction

The United Nations Convention to Combat Desertification (UNCCD) estimated that one-third of the Earth's land surface is vulnerable to land degradation, with more than 1 billion people being at risk of experiencing the effects thereof (Millennium Ecosystem Assessment, MEA, 2005; World Meteorological Organization, WMO, 2005; Low, 2013). In fact, around 73% of the world's rangelands have already deteriorated to such an extent that an estimated 25% of their animal carrying capacity has been lost (Harrison *et al*., 2000; UNEP, 2006).

The effect of land degradation is more severe in drylands, as these systems are especially vulnerable to over-utilisation, climate change, inappropriate land use and the incorrect use of fire (Verón *et al*., 2006). Thus it could be said that land degradation not only results from fluctuations in climate and climatic impacts such as droughts but also from anthropogenic activities, such as mismanagement and the over-exploitation of natural resources (UNCCD, 1994; Hoffman & Ashwell, 2001).

The term drylands applies to arid lands characterised by a low mean annual precipitation (MAP) of less than 250 mm and semi-arid lands that receive between 250 mm and 500 mm MAP (Koundouri *et al*., 2006; EEA, 2012). In the Millennium Ecosystem Assessment, drylands are described as hyper-arid, arid, semi-arid and dry sub-humid areas characterised by evaporation rates that are at least 1.5 times greater than the MAP (UNEP, 2006; Safriel, 2009).

Land degradation is a complex global environmental problem progressively on the increase, ultimately resulting in the temporarily or permanent loss of ecosystem functions (Stocking & Murnaghan, 2001; Schwilch *et al*., 2012). The varying degrees of degradation are particularly important for the African continent, since the UNCCD estimates that two-thirds of the continent can be classified as drylands where most

degradation takes place (WMO, 2005; Whitfield & Reed, 2012). In South Africa, approximately 65% of the rangelands are situated within arid and semi-arid regions (Schulze, 1979; Snyman, 1998; Tainton & Hardy 1999), and almost 25% of these natural rangelands are already degraded (Hoffman & Ashwell, 2001; Kellner *et al*., 1999).

Furthermore, the savanna ecosystems found in South Africa's arid regions are subject to unpredictable rainfall events. Consequently, the erratic moisture supply could give rise to unpredictable fluctuations in plant production, vegetation distribution and composition and basal cover (Snyman & Fouche, 1991; Thomas & Shaw, 1991; Tainton & Hardy, 1999; Sullivan & Rohde, 2002). Added to this, poor grazing management systems, an increase in atmospheric carbon dioxide concentrations, suppression of wild fires and the over-utilization of rangelands for extended periods can decrease the ecosystem's resilience and could result in profound habitat transformations (Polley, 1997; Ibáñez *et al*., 2007).

Due to the reasons mentioned above, savanna ecosystems are particularly threatened by a temporary or permanent imbalance in the grass-woody ratio resulting from mismanagement (Kgosikoma *et al*., 2012). In particular, over-utilisation of rangelands by livestock and game could result in the total removal of grasses. With less grass cover, the woody plants are then able to utilise the greater percolation of water, resulting in a form of land degradation known as bush thickening or -encroachment (Hoffman & Ashwell, 2001; De Klerk, 2004; Thomas & Twyman, 2004). Bush encroachment refers to the invasion of either alien invasive woody plants (e.g. *Prosopis* spp.) or the invasion of indigenous woody plants into environments where the plants did not occur historically. The term bush thickening refers to the increase in density of indigenous woody plants (such as *Acacia mellifera*, *Dichrostachys cinerea* and *Rhigozum trichotomum*) in areas where the woody plants would naturally occur. (In this thesis, the bush thickening problem in southern African savannas is discussed in greater detail in section 2.3.2 of the chapter titled "Literature review").

In 1960, the seriousness of shrub thickening was recognised in the semi-arid savanna Molopo ranching area (present study area) situated in the Mafikeng, Vryburg and Kuruman districts of the North-West Province, South Africa (Donaldson,

1967; Donaldson & Kelk, 1970; Moore *et al*., 1985). Donaldson and Kelk (1970) emphasised that woody plant thickening and drought conditions in this region have a synergistic effect on reducing the grass biomass production. It was found that shrub thickening is capable of decreasing the grass biomass production by as much as 80% within the Molopo ranching area (Moore *et al*., 1985). However, plots cleared of all encroaching woody plants showed a reasonably high grass biomass production during the drought period of 1965/66 (Donaldson & Kelk, 1970). The most important encroacher woody species in the Molopo ranching areas are *Acacia mellifera, Dichrostachys cinerea, Grewia flava* and *A. luederitzii* (Richter *et al*, 2001).

The underlying process of bush thickening and the associated replacement of palatable grasses with unpalatable ones furthermore result in a decrease in biodiversity, rangeland productivity and grazing capacity (Richter *et al.,* 2001; De Klerk, 2004; Smet & Ward, 2005). The latter has significant socio-ecological implications for land users in the arid savannas, since they are forced to apply active or passive actions to improve rangeland conditions, to compensate for the loss in grass forage and to increase the economic value of their lands (De Klerk, 2004; UNEP, 2006; Van Andel & Aronson, 2006; Kellner, 2008). Usually, the removal of woody plants would result in an increase in grass forage production and, thus, in grazing capacity; however, the effectiveness thereof varies from vegetation type to vegetation type (Teague & Smit, 1992).

Against this background, a definite need exists in South Africa for an improved information base to help inform land users on sustainable rangeland management practices (Barac *et al*., 2004; Von Maltitz, 2009). Such an information base should best be construed by following an integrated, participatory approach that combines local knowledge with scientific expertise, while local land users affected by rangeland degradation also ought to be actively involved in the evaluation, decision making and execution processes (Fraser *et al*., 2006; Reed *et al*., 2006; Stringer & Reed, 2007). Furthermore, by implementing a social learning process, the way in which individuals perceive and respond to rangeland degradation can also be influenced positively (Reed *et al*., 2006). In essence, a social learning approach encourages communication and the sharing of knowledge between the local farmers and rangeland scientists with a view to the development of strategies in response to rangeland degradation (Reed *et al*., 2006; Stringer & Reed, 2007).

1.1.1 The PRACTICE approach

The multinational EU-funded project PRACTICE (Prevention and Restoration Actions to Combat Desertification: An integrated approach; www.ceam.es/practice) attempted to narrow the gap that generally exists between land users and scientists by suggesting that a bottom-up approach be followed based on the participatory and integrated evaluation of local-level rangeland management systems and restoration actions aimed at combating rangeland degradation (Rojo *et al*., 2012). As a result, a multi-step participatory, integrated assessment protocol (IAPro) has been developed and subsequently tested at selected dryland sites distributed across 12 countries worldwide. One of the primary objectives of this protocol is to promote social learning through the exchange of knowledge, thereby integrating expert scientific and local knowledge and assessments to capture biophysical and socio-economic information (Bautista & Orr, 2011). In so doing, society and rangeland sciences combine in the process of evaluating and potentially improving implemented restoration and management actions (Bautista & Orr, 2011).

The PRACTICE project commenced in the year 2010 and was executed at several monitoring sites in Chile, China, Greece, Israel, Italy, Mexico, Morocco, Namibia, Portugal, South Africa and Spain as well as in the United States of America. With reference to South Africa, the PRACTICE approach was applied in two different cultural and biophysical settings within the Kalahari farming area, the first area falling within the municipal region of Mier in the Northern Cape Province and the second within the Kagisano-Molopo municipal area in the North-West Province. The study being presented here reports on the application of the PRACTICE approach in the latter, i.e. the bush-thickening prone Molopo rangelands of the semi-arid savannas in the North-West Province.

Rangeland sciences have gained valuable information regarding the processes and drivers of rangeland degradation, but it lacks the incorporation of local indigenous knowledge which can help to find solutions to combat the problem (Stinger & Reed, 2007). Consequently, by following the PRACTICE approach, land users were encouraged to participate in the study with a view to stimulating a mutual exchange of knowledge and experiences amongst scientists and participants to facilitate the enhancement and expansion of both parties' knowledge base regarding rangeland

degradation and appropriate restoration actions. In addition, the Integrated Assessment Protocol (IAPro), which forms part of the PRACTICE programme, was applied to systematically evaluate the local indigenous knowledge as well as the restoration actions implemented by local land users (www.ceam.es/practice/).

1.2 Study objectives

The main objectives of this study were to:

(1) Test the integrative bottom-up approach (i.e. the IAPro) which evaluates the suitability of locally applied restoration and management actions for the mitigation of rangeland degradation in the Molopo semi-arid savanna of South Africa.

Objective 1 would be achieved by:

(a) Identifying restoration and rangeland management actions implemented by local land users and communities affected by rangeland degradation;

(b) evaluating the performance and acceptance of these restoration actions in an integrative manner;

(c) sharing knowledge and results with a multi-stakeholder platform (MSP); and

(d) fostering the implementation of best actions in a locally contextualised manner.

(2) Evaluate the effect of two chemical bush control actions as well as rotational grazing management on the Molopo bushveld vegetation by assessing:

(a) how the woody density affects the frequency distribution of grass and woody species and grass functional groups (GFGs);

(b) the effect of chemical control vs. no control on the productivity parameters of the vegetation; and

(c) whether there is a relationship between the status of the grass layer and woody layer.

1.3 Dissertation structure and content

The dissertation consists of six chapters. The present chapter provides a general introduction and simultaneously outlines the study objectives. Chapter 2 contains a literature review of the land tenure systems of the Molopo farming area and provides an overview of rangeland degradation and the need to combat degradation by way of sustainable grazing management and restoration actions. Also discussed in chapter 2 is the integration of local (indigenous) knowledge with scientific knowledge.

Chapter 3 provides a description of the study area in terms of its location, land use, climate and vegetation. The PRACTICE participatory process and the results obtained by way of the IAPro are discussed in Chapter 4. In Chapter 5, the effect of chemical bush control actions (hand control and aeroplane control) and rotational grazing management on the Molopo bushveld vegetation is evaluated.

Chapter 6 concludes the study and provides recommendations based on the results from Chapter 4 and 5. A complete list of references and appendix is included at the end of the thesis. Note that although raw data is not listed in the appendix, it is available on request.

2

Literature Review

2.1 Land tenure and rangeland management systems in the south-eastern Kalahari

The carrying capacity of grazing ecosystems will largely be determined by the productivity of the grass and woody layer (Fynn, 2012). Abiotic factors such as rainfall, fire and soil fertility are determinants of grass productivity and quality, and biotic factors such as grazing pressure will affect the total herbivore biomass an ecosystem can support (Fynn, 2012). Danckwerts *et al.* (1993) describe rangeland management strategies as the process whereby land users examine possible consequences of various management strategies and then implements the practice that has the best chance of attaining their objectives. The low precipitation, high variability in climate and the soils characteristic of the Kalahari have a great impact on land use, making cultivation of crops very risky as it produces low yields (Jacobs, 2000; Thomas, 2002). Accordingly, livestock ranching and game farming predominate the commercial agricultural sector of the arid to semi-arid Kalahari, while subsistence farming is based on small-stock husbandry (Thomas & Twyman, 2004).

Due to the unpredictability of the seasonal climate characteristic of the semi-arid Kalahari, a management strategy that is adaptive to make ecologically wise decisions that are of economic benefit is needed (Quaas *et al*., 2007). In response to this, different management strategies evolved, some adapted from equilibrium systems based on Clements' successional model (Clements, 1916). This equilibrium model claims that biotic feedbacks such as grazing pressure from livestock densities are seen as the main driver of change in vegetation composition, cover and productivity (Vetter, 2005). Related management strategies thus revolve around an adjustable carrying capacity, range condition assessments and low constant stocking rates with extended resting periods for vegetation recovery (Lamprey, 1983; Dean &

MacDonald, 1994). Carrying capacity is defined as the potential of an area to support livestock and/or game through the grazing and browsing production over a defined period without the deterioration of the overall ecosystem. It may vary from season to season or year to year on the same area due to fluctuating grass forage and woody fodder production (Danckwerts, 1981; Trollope *et al*., 1990). Results discussed throughout the thesis referred to grazing and browsing capacity and not carrying capacity.

In contrast, non-equilibrium systems are primarily controlled by various stochastic abiotic factors, such as droughts (Vetter, 2005), while Westoby *et al*. (1989) consider the high rainfall variability to be the primary driver for vegetation dynamics and claimed that grazing pressure from livestock only plays a marginal role in rangeland condition. Variable rainfall would, therefore, result in highly variable forage production and, accordingly, carrying capacity (Vetter, 2005). Less available forage results in higher mortality rates of livestock or more livestock being marketed, resulting in lower grazing pressure that can be sustained over a longer period, leading to less rangeland degradation. Studies by Illius and O'Connor (1999, 2000), Briske *et al*. (2003), Vetter (2005) and Bashari *et al*. (2008) suggest that both rainfall and grazing/browsing impacts determine vegetation dynamics. With a fluctuating resource base, livestock populations and management strategies aligned with equilibrium systems are rarely capable of reaching a state of equilibrium with negligible effects on the vegetation layer (Vetter, 2005). Semi-arid rangelands include elements from both equilibrium and non-equilibrium models, and the management of these rangelands would, therefore, need to incorporate temporal variability and spatial heterogeneity at different scales (Vetter, 2005).

Land tenure types in the south-eastern Kalahari include commercial-, lease-, open-access communal and subsistence management systems (with mainly small and large livestock) and commercial game management systems, which are addressed separately in Sections 2.1.1 and 2.1.2. The increased use of the Kalahari ecosystem as a rangeland resource is mostly due to the exploitation of ground-water aquifers through boreholes, providing year-round water for human and livestock use (Thomas & Shaw, 1991; Thomas, 2002; Scholes, 2009). Sedentary livestock management systems are, consequently, on the increase in the south-eastern Kalahari savanna regions (Thomas & Twyman, 2004). Definite changes in plant community

composition and structure as well as the condition of the rangeland will occur unless these systems are properly managed and not over utilised (Thomas & Twyman, 2004).

Lease tenure systems are granted by the government to certain individuals, and they hold the rights to use some parts of the land under leasehold (Adams *et al*., 1999). The land holders of lease tenure systems do not own the land, but they can erect fences to exclude others as they have exclusive rights to their leased holdings (Adams *et al*., 1999). Most grazing land not under leasehold is used communally as open-access communal tenure systems as granted to a community by government. Whole communities therefore hold the rights to use parts of the land for grazing for their livestock and the utilisation of other natural resources, for example as fuel and for construction (Adams *et al*., 1999; Nkambwe & Sekhwela, 2006). Most of the commercial rangelands (i.e. land in statuary tenure) were allocated to European-descent settlers during the colonial era as land holdings, and many of these rangelands are, therefore, still under white ownership (Von Maltitz, 2009). These tenure systems are addressed separately in the following section.

2.1.1 Open-access communal and subsistence tenure systems

Communal-managed rangelands can be described as open access to all members of the community living in the area, where natural resources are utilised and also managed communally (individuals do not have freehold titles over the land) (Scholes, 2009; Von Maltitz, 2009). This tenure system has been influenced by the colonial rule for many centuries and is a combination of various traditional tenure systems; however, the communal tenure system of today may have very little in common with pre-colonial communal tenure systems (Von Maltitz, 2009).

Even though the rangeland may be held as a communal resource, individuals may be held responsible to conduct and manage the natural resources, especially for agriculture practices. The legal status of communal land differs within each sub-region, but in most southern African countries, traditional leadership through local chiefs or tribal authorities still plays a vital role in the allocation and management of the land (Von Maltitz, 2009). The majority of rural populations in southern Africa are supported by communal lands, many of which are living below the poverty line (Shackleton *et al*., 2000; Scholes, 2009; Von Maltitz, 2009).

Communal rangelands are largely subsistence-based where meat production is only a secondary objective for sale on local and informal markets (Scholes, 1997; Everson & Hatch, 1999; Wessels *et al*., 2004; Scholes, 2009). Livestock are mostly kept as a form of savings for financial security and also for other products such as milk or hide production (Scholes, 1997, 2009; Everson & Hatch, 1999; Shackleton *et al*., 2005; Twine, 2013). Land users on communal rangelands are largely sustained financially by income earned outside the agricultural rural areas, which include government subsidies and transfers (e.g. pensions and an array of grants) (Everson & Hatch, 1999; Baker & Hoffman, 2006; Scholes, 2009). Although the communal-managed areas are capable of keeping half of the livestock population in South Africa, their production is likely to be much lower (Everson & Hatch, 1999; Scogings *et al*., 1999).

The historical, climatic and socio-economic context of communal rangelands need to be considered if sustainable rangeland management systems for these unique conditions common to communal areas are to be developed (Hoffman & Ashwell, 2001). For many decades, the movement of livestock on a seasonal basis was part of the livestock management system in communal areas. African pastoral societies would have responded to droughts by retaining a high degree of mobility and moving livestock to areas less affected by the drought conditions, thus having higher forage availability (nomadic pastoralism) (Scoones, 1993; Everson & Hatch, 1999). Livestock were then left in the care of relatives or hired managers, in exchange for the use of livestock products such as milk. This management system implemented by the pastoralist to reduce livestock mortality during extreme drought periods is unique to southern Africa and is called "*mafisa"* (Thomas, 2002; Hitchcock, 2002 *in* Reed *et al*., 2007). Although "*mafisa"* is still implemented in some areas, the privatisation of grazing lands, increase in human populations and political changes add a limitation to the available grazing reserves that could be utilised during drought periods (Baker & Hoffman, 2006; Reed *et al*., 2007).

Today, herding of livestock on communal rangelands is only seen as an opportunistic management system which is still used to avoid droughts and to herd livestock in areas with increased available forage (Scoones, 1993; Baker & Hoffman, 2006). Without any fences on communal rangelands to make camps and implement rotational grazing to ensure the recovery of the vegetation after being utilised, most

of the areas are overgrazed. Herding of livestock is an efficient management strategy implemented in drylands with highly variable precipitation and forage availability (Scoones, 1993; Baker & Hoffman, 2006). There are two contrasting herding strategies: (1) the sedentary herding strategy that makes use of a primary livestock post (small fenced camp where livestock is kept during the night to protect the livestock from predators and theft) with little or even no movement between different livestock-posts throughout the year, and (2) the mobile herding strategy where herders move the livestock posts at least once a year (Baker & Hoffman, 2006). Communal and lease farmers tend to keep goats as well since the feeding habits of these animals allow them to graze and browse (i.e. intermediate feeders), making them more drought tolerant than sheep or cattle. Smaller livestock also tend to have a better recovery rate following a drought period compared to larger livestock such as cattle (Peacock, 2005; Reed *et al*., 2007).

Various perceptions around the management of communal rangelands exist, such as that these rangelands are degraded, non-sustainable and have a low agricultural production (Hardin, 1968; Sinclair & Fryxell, 1985; Everson & Hatch, 1999; Hoffman & Todd, 2000; Wessels *et al*., 2004; Vetter *et al*., 2006; Palmer & Bennett, 2013). It is, however, not the communal livestock management system *per se* that leads to rangeland degradation, but rather that no management over the natural resources exists, allowing open access to all members of the community with very little control over the size and movement of the herds (Von Maltitz, 2009).

Neither the government nor the banks and not even farmers want to invest in communal livestock rangelands due to their low productivity and the ineffectiveness of control over resources. The Amalgamated Bank of South Africa (ABSA, 2003) even reported that game ranching offers a better solution for communal rangeland that are mostly associated with livestock diseases, theft and competitive agricultural markets, yet none of the established game farms are in former homelands (Carruthers, 2010).

According to Scholes (2009) and Vetter (2013), the lack of sustainable rangeland management on communal land is mainly due to landlessness, little ownership and control over the land, an increase in local human population, social stratification, conflicting interests of the local community members, local power struggles, political

and ethnic divisions, theft of livestock and a lack of credible local institutions. Traditional grazing arrangements have also been disrupted by the resettlement of members from other surrounding communities into the already overcrowded community, resulting in divisions of management committees and creating conflict over the use and management of natural resources (Vetter, 2005; 2013). Sustainable agricultural land-use on communal and lease rangelands is, however, vital for the economic and social development of these areas (Everson & Hatch, 1999).

A need exists for local institutions (e.g. traditional leadership or civil society) to take ownership of the natural resources and to ensure effective management thereof by securing land rights for the local community members (Ainslie, 1999; Vetter, 2013). The relevant policy needs to be informed by the best current ecological, social, political and historical understanding to support local land users and to focus on a small-scale, sustainable agricultural production system and to develop better local-market subsistence livestock keeping practices (Nkonya *et al*., 2011; Vetter, 2013). Local community members should be trained to be made aware of ecologically appropriate rangeland management and monitoring abilities that align with their production objectives, disease control issues and livestock husbandry (Vetter, 2013). This can be achieved by agricultural training colleges with updated curricula to implement suitable training for the land users managing livestock in communal rangelands.

A need for training in alternative management systems also exists, such as how to apply land-use changes from livestock towards wildlife conservation, as well as appropriate policies for sustainable resource use in places that will contribute to human wellbeing, increase livelihood options and improve the benefits gained by the community of the rangeland (Fischer *et al*., 2011; Chaminuka, 2013). Secondary rangeland resources collected on communal rangelands for additional income and use are referred to by Cousins (1999) as hidden capital, such as edible fruits, honey, thatch grass, fuel wood and carving woods. Many studies highlight income gained from secondary resources among poorer households which often makes a greater contribution to the total yearly income per household than the income from livestock sales (Cavendish, 2000; Letsela *et al*., 2002; Thondhlana *et al*., 2012).

2.1.2 Commercial tenure and rotational grazing management systems (multi-camp systems)

Commercial rangelands vary from hundreds to thousands of hectares with the primary land use being livestock production to be sold on formal markets (Scholes, 2009; Von Maltitz, 2009). The commercial freehold land tenure system is often retained by the children of the farmer through inheritance conditions, family rangelands or joint ownership of the rangeland (Von Maltitz, 2009).

Most commercial systems are based on a multi-camp approach allowing the application of rotational grazing management systems. The rotational grazing system allows a recovery period in between grazing periods for better vegetation production over the long term (O'Connor *et al.*, 2010). This management system is most commonly applied by land users in the south-eastern Kalahari and therefore it was decided to focus on this management system within commercial tenure systems.

It was near the end of the 18th century in Scotland when James Anderson described the principle of rotational grazing management (Voisin, 1988), and it is defined as a grazing management system that requires the sub-dividing of the rangeland into smaller enclosures/camps. Accordingly, a group of animals is allotted to a camp and then moved into a next camp after most of the vegetation has been grazed or as decided by the land manager. Vegetation in camps not grazed, depending on the rainfall, will be able to recover before being utilised again (Booysen, 1967; Tainton *et al*., 1999; Briske *et al*., 2011). This involves the continuous grazing of the camps in a rotational manner so that the animals are only concentrated on a small part of the graze-able land (Booysen, 1967; Tainton *et al*., 1999). Other means to implement rotational grazing is the instalment of watering points to distribute the movement of animals equally over larger areas and to provide specific grazing areas with longer periods of rest as the animals migrate between the watering points (Owen-Smith, 1999; Briske *et al*., 2008).

The primary objectives of rotational grazing systems are to (a) control the frequency at which the plants in each camp are grazed, (b) control the intensity at which plants are being utilised by limiting the number of animals allowed in each camp and keeping the animals for a specific time period in the camp and to (c) reduce selective

grazing by confining a relatively large number of animals in a small camp (Tainton *et al*., 1999).

Resting of a camp provides plants with an extended period where no grazing and browsing take place, allowing plant development to complete the necessary phonological processes for plants survival, such as flowering and seed setting and dispersal (Tainton *et al*., 1999). Forage biomass accumulates during this period for livestock grazing and animal production (Tainton *et al*., 1999). Tainton *et al.* (1999) and Snyman (1998) suggested that biomass accumulation is probably the most critical element in any grazing management system as damage caused to plant vigour through grazing impact may be limited without appropriate resting/recovery strategies.

Müller *et al*. (2007) did a modelling analysis on the relevance of rest periods in non-equilibrium, semi-arid rangeland systems. The results from the model showed that resting a third of the camps during years with an above mean annual precipitation is crucial for the regeneration of pastures and ensures new reserve biomass build-up. When resting is applied during the dry years, vegetation barely benefits, as there is not enough available moisture to ensure new reserve biomass build-up (Müller *et al*., 2007). After a drought, the rapid recovery of vegetation can be facilitated by allowing a resting period in the following years (Danckwerts & Stuart-Hill, 1988).

The aim when managing a multi-camp system is to minimise the effect of patch overgrazing (Teague & Dowhower, 2003). The latter prevents large variances in the species composition between palatable and unpalatable plant species and is aimed at maintaining or improving the veld condition. Thus, rotational grazing management provides the land user with the ability to manipulate a non-equilibrium rangeland system in savanna ecosystems (Walker *et al*., 1989; Briske *et al*., 2008, Briske *et al*., 2011). The grass and woody layer can be utilised separately by rotating browser and grazer livestock at different intervals, based on the condition of the two different vegetation types and structures, i.e. lower growing herbaceous and taller woody vegetation components.

Other methods to implement rotational grazing management are with intense grazing in small camps exerting a high grazing pressure on the vegetation for a short period of time in order to limit selective grazing (Quaas, *et al*., 2007), referred to as holistic

rangeland or resource management (Savory & Parsons, 1980; Savory, 1999). This is made possible by small camps which can be around a watering point in a wagon-wheel layout (Savory & Parsons, 1980). The small camps are then stocked with both cattle and sheep capable of utilising the coarse grass material more efficiently (Müller *et al*., 2007). When combined with goats that will also utilise available shrub forage, so-called “herd effects” of concentrated livestock grazing are created (Savory & Parsons, 1980; Savory, 1999; Briske *et al*., 2011). This management system requires higher costs as more fencing for the erection of small camps and more watering points are required (Kellner, 2008; Briske *et al*., 2011).

The time livestock can spend in a camp before being rotated depends on the (a) grass biomass available within the camp, (b) density and total intake of the livestock type, (c) the grass biomass proportion that can be removed, (d) whether rumen stimulants are provided to the grazer and (e) the size of the camp (Beukes *et al.*, 2002; Müller *et al*., 2007). The smaller the camp size, the less time livestock can spend in the camp, reducing the recovery period of the vegetation which will have an effect on when the particular camp can be grazed again.

2.2 General aspects of vegetation dynamics shaping savanna ecosystems

Savanna ecosystems have an herbaceous, usually graminoid understory layer and an upper layer of less continuous woody plant cover, which can vary from widely spaced to a canopy cover of 75% (Rutherford & Westfall, 1986; Trollope *et al*., 1990; Skarpe, 1991; Scholes & Archer, 1997; Rutherford *et al*., 2006). Vegetation structure, composition and the geographic distribution of savanna ecosystems are mainly determined by primary (climate and soil properties) and secondary determinants (herbivory and fire) (Scholes & Archer, 1997; Scholes *et al*., 2002; Sankaran *et al*., 2005; Wiegand *et al*., 2006; Sankaran & Anderson, 2009; Higgins *et al*., 2010). Mean annual precipitation (MAP) and herbivore behaviour are able to regulate the grass biomass production (Wiegand *et al*., 2006; Scholes, 2009). Fire intensity, on the other hand, is controlled by the available grass biomass production and woody plant growth, while regeneration of vegetation is regulated by the fire intensity (Trollope *et al*., 2002; Van Langevelde *et al*., 2003; Wiegand *et al*., 2006).

2.2.1 Climate

Large inter-annual climate variations are largely responsible for changes in the species composition of savanna ecosystems (Scholes, 1997; Holmgren & Scheffer, 2001; Sankaran *et al.,* 2005, 2008; Fensham *et al.,* 2009; Holmgren, 2009; Buitenwerf *et al*., 2011). Relative low annual precipitation and high evaporative losses (high moisture stress) exert major influences on plant survival (Donaldson & Kelk, 1970; Behnke & Scoones, 1993). Sankaran *et al.* (2008) documented MAP to be the most important driver of woody cover. The data indicated a dependence of woody plant cover at a MAP of between 200 and 700 mm, suggesting water limitations to regulate the woody community structure. Above 700 mm MAP, other disturbances reducing woody plant cover and permitting grass plants to coexist are required in order to maintain a savanna ecosystem (Bond *et al*., 2003[a], Sankaran *et al*., 2005, 2008). Accordingly, African savannas seem to switch from rainfall-dependent ecosystems towards disturbance-dependent ecosystems along a rainfall gradient with the transition occurring between 650-700 mm MAP (Sankaran *et al*, 2008).

Much evidence for a positive relationship between MAP and plant diversity exists (O'Brien, 1993; Shackleton, 2000; Scholes *et al*., 2002). In the Kalahari savanna, at above 200 mm MAP, an increase in the woody basal area at a rate of 2.5 $m^2 ha^{-1}$ per 100 mm MAP occurs (Scholes *et al*., 2002). The number of woody plant species contributing to the basal area increases from one at 200 mm MAP to sixteen at 1 000 mm MAP (Scholes *et al*., 2002). The woody layer of regions receiving between 200 mm and 400 mm MAP is largely dominated by *Acacia* and *Dichrostachys* species. It is replaced by broad-leaved species of the genera *Combretum*, *Terminalia* or *Colophospermum* at a MAP of between 400 mm and 600 mm. Above a MAP of 600 mm, the savanna is largely dominated by other representatives of the *Caesalpiniaceae* family (Scholes *et al*., 2002).

The overall trend for savanna ecosystems is for the mean woody biomass, cover, basal area and height to increase along an increasing MAP (Scholes *et al.*, 2002; Scheiter & Higgins, 2009). Since humidity reaches very low values during the day in most semi-arid savanna regions, many plant species are unable to replace the moisture lost by high transpiration rates at such low humidity rates (Schulze, 1979;

Tainton & Hardy, 1999). Plant leaves start to lose turgidity and wilt, resulting in a slower growth and low survival rate. Plants overcome these processes by developing a deciduous habit (i.e. dropping their leaves when water is not readily available) (Rutherford *et al*., 2006) or developing adapted, modified leaves. Smaller leaves have a lower transpiration rate by developing a thick cuticle and fewer pores (stomata) through which water can be lost (Tainton & Hardy, 1999; Ward *et al.*, 2012). The adapted plant species of the *Mimosaceae* family with smaller, bipinnately compound leaves are assumed to occur more dominantly in the hot and drier savanna regions (Scholes *et al*, 2002). Mature and slow growing plants are also adapted to endure such conditions compared to young seedlings and vigorously growing plants (Tainton & Hardy, 1999).

An increase in woody plant biomass may not only be influenced by variable rainfall patterns (Sankaran *et al*., 2005) but also by an increase in the atmospheric CO_2 (Idso, 1992; Polley, 1997; Polley *et al*., 1999; Bond & Midgley, 2000; Ward, 2010; Bond & Midgley, 2012). The adaptive dynamic global vegetation model by Scheiter and Higgins (2009) and the resource ratio model of the effects of changes in CO_2 on woody plant invasion by Ward (2010) predict a substantial increase in woody plant dominance due to climate change and changes in CO_2. Large areas of the savanna systems (45.3%) will be replaced by deciduous woodlands under elevated atmospheric conditions, and it is estimated that 34.6% of grasslands will be transformed into savanna-type vegetation (Scheiter & Higgins, 2009). Elevated CO_2 also reduces transpiration in both C_3 and C_4 grasses (Wand *et al*, 1999; Morgan *et al*., 2001; Stock *et al*., 2005), resulting in an increase in soil moisture availability and increasing the above-ground productivity of the vegetation (e.g. C_3 woody plants) previously excluded due to low moisture availability (Scholes & Archer, 1997; Polley *et al*., 1999; Bond & Midgley, 2000; Morgan *et al*., 2001, 2004; Bond *et al*., 2003[b]; Kgope *et al*., 2010; Tietjen *et al*., 2010).

However, according to Lohmann *et al*. (2012), the studies mentioned above do not take into account the inter-species competition of savanna woody plants (Tietjen *et al.,* 2010) and therefore implies that it may refer to more mesic savanna environments with higher water availability (Kgope *et al*., 2010). The germination and successful establishment of woody plants are mostly determined by the available soil

moisture when CO_2 and mean annual temperature increases occur (Classen *et al*., 2010).

Irrespective of changes in precipitation patterns and elevated levels of atmospheric $CO_{2,}$ the simulated long-term response to climate change of semi-arid rangelands study by Lohmann *et al*. (2012) revealed that the risk of bush thickening to be lower due to future increasing mean temperatures. Encroaching woody plant species are highly sensitive to drought conditions and/or severe winters when it comes to the critical processes of germination and successful establishment (Hatch, 1999; Lohmann *et al*., 2012). Lohmann *et al.* (2012) also highlighted that higher inter-annual variation and decreasing precipitation, together with temperature increases, would result in severe losses of perennial grass biomass.

The fluctuations in water availability during seasonal changes between wet and dry seasons affect the coexistence of woody and grass plants (O'Connor, 1991; Scanlon *et al*., 2005; Guttal & Jayaprakash, 2007; Hassler *et al*., 2010). Perennial grass species are more dominant in the savanna systems with the number of annual species increasing with aridity, as well as in mesic savannas (400 to 1000 mm MAP) in low rainfall years (Scholes, 1997). Absolute grass biomass production increases in southern African savannas along an increasing MAP up to 600 mm, thereafter decreasing due to higher competition from woody plants at the higher MAP levels (Scholes *et al*., 2002). Observations by Scanlon *et al.* (2005) revealed woody plant densities are more dependent on the long-term MAP of the wet season, while the grass cover is dependent on the higher soil moisture availability near the soil surface over the short term (annual basis) as well as the densities of woody plants.

2.2.2 Fire

Fire can play an integral part in the management of savanna ecosystems. It influences the dynamics and structure of the vegetation, often favouring the grass layer over the woody layer (Scholes & Archer, 1997; Trollope, 1999; Higgins *et al*., 2000, 2007; De Klerk, 2004; Sankaran *et al.,* 2004, 2005; Bond *et al.,* 2005; Sankaran & Anderson, 2009; Joubert *et al*., 2012). Fire frequencies in moist savannas (800 to 2 000 mm MAP) can occur annually; however, in the more arid savannas (< 650 mm MAP), fire occurs naturally only once every ten or more years (Scholes, 1997; Donzelli *et al.*, 2013).

The low frequency of fire in semi-arid savannas can be ascribed to the highly erratic rainfall patterns, as fires are often ignited by lighting during thunderstorms. The frequency of fires in rangelands may also be suppressed by fire policies that exist in the region or areas that are overgrazed which reduce the accumulation of grass biomass required for fuel in fire events (Trollope, n.d. *in* De Klerk, 2004; Scholes, 2009; Joubert *et al*., 2012). Fires can also be caused accidentally or even deliberately by human actions. These causes have a considerable influence on Kalahari vegetation communities (Thomas & Shaw, 1991; Scholes *et al*., 2002; De Klerk, 2004). In southern African savannas, fire is more likely to occur towards the end of the dry season when the vegetation is dryer (Thomas & Shaw, 1991; De Klerk, 2004; Rutherford *et al*., 2006; Sankaran & Anderson, 2009).

Arid to semi-arid savannas are more likely to be rainfall-dependent when considering woody plant recruitment, while in moist savannas, fire exerts greater control over the woody cover (Bond *et al.,* 2003[a]; Bond & Keeley, 2005; Sankaran *et al.*, 2005, 2008; Kgope *et al*., 2010). Fire was found to be the second most influential factor for woody cover in high rainfall African savannas by Sankaran *et al.* (2008). Studies conducted by Higgins *et al.* (2000) and Sankaran *et al*. (2005) have shown that in savannas receiving less than 650 mm MAP, fire has a less significant influence on the change in the mean density of woody plants. Fire only affects savanna structure at MAP levels > 820 mm (Higgins *et al.,* 2010). At such high MAP levels, fire becomes a necessity for grasses to be able to persist within the system (Higgins *et al.,* 2010).

Fire is able to increase the forage quality and the re-growth of plants greatly (Fynn, 2012). Re-growth after a fire is important for pre-existing vegetation over the short term (Scholes, 1997). However, the exclusion of fire over the longer term does lead to changes in the plant species composition in both the more moist and more arid savanna systems. Species such as *Panicum aequinerve and P. maximum* that are more fire sensitive and shade tolerant tend to colonize these areas (Scholes, 1997; Tainton & Hardy, 1999; Smit, 2004; Masunga *et al*., 2013).

Plants in east-African savannas have shown higher levels of nitrogen (N), phosphorus (P), potassium (K), calcium (Ca) and magnesium (Mg) after a fire compared to vegetation that was left undisturbed by fire (Van de Vijver *et al*., 1999). Nutrients are lost into the atmosphere due to fire, but the advantages of this is that

fire oxidises organically bound elements in the vegetation and dead organic material, releasing them in forms that can then be filtrated back into the soil, thereby promoting the growth of new shoots (Thomas & Shaw, 1991; Badejo, 1998; De Klerk, 2004; Rutherford *et al.*, 2006; Kgope *et al*., 2010).

Woody leaf biomass of trees smaller than 2 m located within the flame zone can be reduced by frequent fires. The so-called top-killing of seedlings and saplings keeps the woody plants in a juvenile state (Van der Walt & Le Riche, 1984; Thomas & Shaw, 1991; Scholes, 1997; Higgins *et al*., 2000; Bond *et al*., 2003[a]; De Klerk, 2004; Bond & Keeley, 2005; Sankaran *et al.,* 2008). Fire intensity is highly variable, depending on the accumulation of available fuel material and structure of the vegetation, the moisture content thereof, wind speed and atmospheric humidity (Teague & Smit, 1992; Trollope, 1999). Woody saplings are able to escape the deadly zone of fire and grow to a suitable size (taller than 2 m), making them fire-resistant, especially when the occurrence of fire is rare (Higgins, 2000; Bond *et al*., 2003[a]; Bond & Keely, 2005; Sankaran, 2008; Scheiter & Higgins, 2009).

All factors mentioned above largely prevent shrub thickening in semi-arid savannas (Joubert *et al*., 2012) as the occurrence of fire is too infrequent. Surface fires (i.e. fires burning on surface fuels, including standing grass, small shrubs, forbs and fallen leaves and twigs) normally burn as either head- or back-fires (Trollope, 1999). A back fire or a cool-head fire is not hot enough to kill the woody plants and can, as a consequence, promote shrub thickening since the fire will only remove the grass layer and provide more soil moisture for the woody layer after the removal of the grass (Trollope, 1999; De Klerk, 2004).

The grass fuel load required to ensure a hot fire that will control woody plant growth and reduce the woody density in southern Africa is 4 000 kg/ha or higher (Trollope, 2011). In the semi-arid savannas, these high fuel loads are rarely found in rangelands (Joubert *et al*., 2012), especially in the areas were shrub thickening has already occurred and the grass biomass is too low (Scholes, 2009). Tree densities can, therefore, increase immediately after a once-off fire since the scarification of the seeds could result in stimulating the germination of woody seedlings (De Klerk, 2004) and the creation of suitable open spaces for seed germination.

Moribund material accumulating from the grass layer is removed by fire, which may create open patches for the establishment of species-rich communities (ranging from smaller annual, pioneer to sub-climax type species) that are normally suppressed by more dominant perennial, climax species (Scholes, 2009; Masunga *et al*., 2013). Rare species, however, have a low tolerance for fires and could also become extinct when fires occur too frequently (Masunga *et al*., 2013). Fires also tend to reduce fire-sensitive species. For example, the woody plant species *Boscia albitrunca* is more fire resistant than the fire-sensitive *Acacia* species.

On a landscape scale, Zimmermann *et al.* (2010) explored the influence of fire on tuft mortality of the perennial, sub-climax grass *Stipagrostis uniplumis*. They found that although the amount of accumulated moribund material increased grass tuft mortality rates, fire indirectly reduced future mortality of grasses by decreasing the shading effect of the large tufted grass as well as competition from other grasses (Zimmermann *et al.*, 2010). Therefore, the frequency and timing of fires are crucial to ensure effective plant growth (Zimmermann *et al.*, 2010).

The general attitude towards fire as a veld management tool in semi-arid savannas tends to be negative, except in conservation areas (Trollope, 1999). Burning the veld deliberately has been carried out as an element in hunting strategies in the past and as a grazing management strategy, mainly to (a) stimulate and promote the re-growth and quality of grass species out of season, (b) remove accumulated moribund material and (c) to control of shrub thickening or -encroachment (Thomas & Shaw, 1991; Trollope, 1999; Hoffman & Ashwell, 2001). A study conducted by Bester (1996) showed that only between 15% and 25% of woody plants are killed by fire and that most species coppiced and were able to regenerate and grow actively after fire events. To a degree, fire only succeeded in reducing the growth rate of woody plants and, to a lesser degree, to suppress the establishment of woody saplings. Thus, fire should rather be used as a preventative measure and not as a tool to clear large woody shrub-encroached areas (De Klerk, 2004).

2.2.3 Herbivore impact

It is believed that all herbivore species control the structure of savannas and can affect the primary productivity and rates of nutrient cycling both positively and negatively (Forana & Du Toit, 2007; Sankaran *et al*., 2008). This is true for large

grazing mammals, as well as browsing ungulates and mixed feeders (O'Connor, 1994; Van Langevelde *et al.*, 2003; Augustine & McNaughton, 2006; Bobe, 2006; Allred *et al*., 2012). Herbivores have both direct (browser species decreasing the woody cover) and indirect effects (reducing biomass production of grasses which, in turn, reduces the frequency of fires that have an impact on the woody layer) on the vegetation structure (Skarpe, 1991; Van der Waal *et al*., 2011).

Another indirect effect of heavy browsing is that more light is able to penetrate the woody canopy, affecting the herbaceous species composition (i.e. more less shade-tolerant species are able to survive) underneath the canopy (Allred *et al*., 2012). If the grazing pressure is regulated and followed up by long resting periods, the vegetation may regenerate, which may stimulate the growth of the more palatable, climax-type grass species, suppressing woody plant growth (Fynn & O'Connor, 2000; Vetter, 2005; Riginos, 2009; Riginos *et al.*, 2009; Scholes, 2009). The latter will depend on the amount of rainfall.

In the past, prior to the introduction of firearms in southern Africa, heavy defoliation by large migratory herds of herbivores reduced the available plant resources needed for recovery after stress and for the investment in a better root system (Scholes, 2009; Rohde & Hoffman, 2012). Grass plants adapted a more rapid lateral growth form in order to survive herbivory from larger, taller-standing mammals (Ingrouille & Eddie, 2006; Forana & Du Toit, 2007). At larger spatial scales, vegetation characteristics (e.g. palatability and defence mechanisms) can influence the selection made by herbivores to forage as the surrounding matrix of each site can be attractive (palatable) or repellent (less palatable) for the herbivore (Baraza *et al*., 2006; Ingrouille & Eddie, 2006; Forana & Du Toit, 2007; De Knegt *et al.*, 2008). In nutrient-poor savannas, plants adopted a chemical defence against browsing (tannins, resins and polyphenolics); whereas in nutrient-rich savannas, structural defence mechanisms (e.g. spines) were formed (Scholes, 1990; Rohner & Ward, 1997; Scholes *et al*, 2002; Ingrouille & Eddie, 2006; Rutherford *et al.*, 2006).

The selective consumption behaviour of herbivores in savanna systems results in spinescent woody and unpalatable grass species dominating the system (Hardin, 1968; Augustine & McNaughton, 1998; Fynn & O'Connor, 2000; Augustine & McNaughton, 2006; Palmer & Bennet, 2013). These systems were sustainable as

the herbivores responded to the changes in the vegetation (e.g. the available palatable plant material declined) by migrating, allowing the plants time to recover (Skarpe, 1991; Scholes, 2009). According to the simulation model by De Knegt *et al.* (2008) regarding the role of herbivore densities as a determinant of changes in vegetation patterns, in the past, higher densities of migrating browsers could have resulted in an increase in browsed patches up to a point where almost all woody vegetation had been utilised and only scattered areas of un-browsed vegetation were left, thus creating open grassland savannas with scattered woody plant cover.

Today, fences, permanent water, the settling of nomadic populations and artificial feeding of livestock prevent the spatial response to temporal variability (Scholes, 2009). Ultimately, frequent defoliation for a long period of time and continuous movement of ungulates will result in the loss of vegetation cover and a reduction in the efficiency of converting precipitation and radiant energy into plant biomass (Scholes, 2009). Herbivory at high densities for a long period of time may reduce the primary productivity of the ecosystem, over-utilise highly palatable species and inhibit nutrient cycling rates (Augustine & McNaughton, 2006; Sankaran *et al*., 2008; Scholes, 2009). The system will, therefore, become dominated by unpalatable plant species with a slower growing rate, which will cause herbivores to move away (Hardin, 1968; Scholes, 1997; Fynn & O'Connor, 2000; Augustine & McNaughton, 2006; Sankaran *et al*., 2008; Scholes, 2009).

However, systems with large herbivore populations for a very short period of time have shown that herbivory can also stimulate primary productivity, nutrient cycling rates and secondary productivity (Augustine & McNaughton, 1998; Augustine & McNaughton, 2006). The grass production of broad-leaved *sourveld* savannas is high; however, a small proportion of the grass layer is utilised, and as the grasses reach maturity, they become less palatable and also less nutritious to the animals during mid- to late summer (Hardy *et al*., 1999; Stuart-Hill & Tainton, 1999; Scholes, 2009), therefore the southern African term *sourveld.* In contrast, fine-leaved *sweetveld* savannas in the semi-arid regions as represented by the south-eastern Kalahari ecosystems produce less grass biomass, but a larger fraction may be consumed since perennial grasses are palatable year-round to grazing animals resulting in a higher secondary production (Hatch, 1999; Stuart-Hill & Tainton, 1999; Scholes, 2009).

Trampling and the uprooting of plants as a result of the grazing patterns of herbivores maintain the diversity within savanna systems (Skarpe, 1991; Scholes, 1997). Seedlings and saplings of woody plants are particularly vulnerable to the herbivory from browser and grazer ungulates, since they are also affected negatively if consumed during grazing (Huntley, 1991; Hoffman & Ashwell, 2001; Van Langevelde *et al.*, 2003). Woody plant seedlings growing within the clumps of unpalatable grass species may, however, escape herbivory (Scholes & Archer, 1997).

2.3 Degradation of southern African savannas

2.3.1 Overview of processes and current state

The United Nations Environment Programme claims that 10 to 20% of the world's drylands are already degraded (UNEP, 2006). Africa is impacted most with 73% of its agricultural drylands being degraded (Hoffman & Ashwell, 2001). By definition, land degradation is a physical process characterised by the loss of biological and/or economic productivity within an area. Serious land degradation processes will result in desertification, especially in arid, semi-arid and dry, sub-humid regions (UNCCD, 1994; Levia, 1999; Hoffman & Ashwell, 2001; Safriel, 2009; Palmer & Bennet, 2013).

One-third of the earth's surface and more than one billion people are affected by land degradation (UNEP, 2006). Carbon-, hydrological- and nutrient cycles are affected negatively by land degradation, which may be critical for human survival (Milton *et al*., 1994; Tongway & Ludwig, 1996; Bossio *et al*., 2010; Cowie *et al*., 2011; Palmer & Bennet, 2013). Land degradation also contributes to soil loss, water and air pollution, biomass and bio-productivity loss and the total amount of dust and carbon dioxide in the atmosphere (Milton *et al*., 1994; Cowie *et al*., 2011). This has a large effect on ecosystem productivity, food security, national economic development and natural resource conservation strategies (Wessels *et al*., 2007). The degradation of rangelands also involves the loss of or shift in species composition due to changes in habitat conditions (Ayyad, 2003; Palmer & Bennet, 2013).

Globally, climate change, poverty and food insecurity greatly affects the way in which land users manage and utilise their land (Snel & Bot, 2000; Stocking & Murnaghan,

2001). Poverty and higher socio-economic pressures result in the overutilisation of natural resources, decreasing the ability of the environment to restore lost resources as it requires too much labour and capital investment by the land user (Meadows & Hoffman, 2002; Von Maltitz, 2009). It also results in land users tending to focus on more immediate needs, such as education and medical needs, rather than on those that would benefit the rehabilitation of the degraded rangelands in the long term (Stocking & Murnaghan, 2001).

Land users who own the land (mostly commercial farmers that already have a good infrastructure for rangeland management) are more likely to consider investing money and labour in the rehabilitation of natural resources because they realise the benefits of better vegetation and animal production may result in a positive feedback over the long term (Snel & Bot, 2000; Stocking & Murnaghan, 2001). Conversely, communal-managed rangelands (often with poor infrastructure for better rangeland management) with open access to all the land users have free rein to make use of any resources, and control over these resources is limited (Stocking & Murnaghan, 2001). Poor societies in southern Africa usually have the least secure land tenure and/or restricted resources and land, decreasing their ability to manage the land sustainably (Beinart, 2000; Snel & Bot, 2000; Meadows & Hoffman, 2002).

The encroachment and thickening of indigenous and exotic woody shrub and tree species in a Savanna ecosystem, outcompeting productive herbaceous forage, is theoretically reversible over a short timeframe. However, socio-economic constraints may cause the change to be effectively permanent (Stocking & Murnaghan, 2001), as many land users do not have the economic incentives to remove these woody species. This contributes to the perception that rangeland degradation is an anthropogenic phenomenon, which cannot only be ascribed to natural processes such as climatic variations (Stocking & Murnaghan, 2000; Reed, 2005; Hudson & Alcntára-Ayala, 2006).

Following on severe droughts in sub-Saharan Africa, international donor organisations invested millions of dollars to (1) improve rangeland production, (2) install watering points in order to settle nomads and (3) to prevent rangeland degradation taking place due to overgrazing (Sinclair & Fryxell, 1985; Oba *et al*., 2000[b]). Despite all the money invested in many regions throughout Africa, countless

grazing projects have failed in general, especially in communal-managed rangelands, as more urgent needs (e.g. sanitation, medical and education facilities) had to be addressed. The main issue was that grazing projects changed traditional patterns of land use, resulting in severe rangeland degradation and economic decline (Oba *et al*., 2000[b]).

General grazing management systems implemented within equilibrium systems have failed when applied in dryland regions. Furthermore, the development of watering points for livestock use and the exclusion of grazing (e.g. implementing a resting period) altered traditional land-use patterns since higher livestock numbers were obtained, an outcome the financially burdened land user is likely to view as an asset that can be used to address more urgent needs. These changes in land use resulted in rangeland degradation and a loss of forage production (Sinclair & Fryxell, 1985; Oba *et al*., 2000[b]).

Zucca *et al*. (2011) analysed the drivers and related problems of rangeland degradation most widely reported by stakeholders affected on a regional to global scale. They found that the drivers that lead to land degradation may be a combination of factors, such as (a) the overutilisation of agricultural land, unsustainable rangeland management and increased soil erosion; (b) overuse of water resources with intensive irrigation resulting in salinisation; (c) mismanagement of grazing practices, overgrazing of rangelands, sand or shrub encroachment and reduction in herbaceous forage production; (d) deforestation; (e) increased aridity; (f) changes in population distribution, increasing populations and land abandonment, all leading to socio-economic problems, and (g) expansion of mining and industrial activities, leading to extensive air and water pollution and soil loss by contamination.

The causes of rangeland degradation need to be identified and thoroughly addressed in the early stages to ensure that the land and ecosystem function is restored and to prevent an increase in poverty resulting from these processes (Von Maltitz, 2009). Land users should also become more aware of and more effectively involved in the process of combating rangeland degradation (Reed *et al*., 2007; Von Maltitz, 2009). This can be achieved through integrative studies making use of site-specific indicators identified by local land users most affected by rangeland degradation.

However, Reed *et al*. (2007) claim that the involvement of local land users in southern Africa (especially Botswana) and globally in the assessment of rangeland degradation is both rare and passive. According to these authors, another issue would be the application of top-down approaches followed by the implementation of scientific knowledge in rangeland degradation solutions without integrating the various components of rangeland degradation, i.e. bio-physical and socio-economic. This often prevents rangeland scientists from seeing the multifaceted nature of the degradation problem (Reed *et al*., 2007).

2.3.2 The phenomenon of woody shrub and tree thickening

The following section will pay special attention to the land degradation syndrome 'shrub thickening' (Scholes, 2009) due to its severity in southern African savanna rangelands, including the semi-arid parts of the Kalahari being addressed in this study.

Shrub thickening or the proliferation of indigenous woody species in the semi-arid savanna regions is mostly ascribed to overgrazing of herbaceous plants by livestock as well as fire and drought (Trollope, 1980; Scholes & Archer, 1997; Kgosikoma *et al*., 2012). This phenomenon can be defined as the invasion or thickening of aggressive and undesired woody plant species that causes an imbalance in the ratio of grass to bush, decreasing agricultural production and causing a loss in biodiversity (Oba *et al.,* 2000[a]; Hoffman & Ashwell, 2001; Scholes, 2003; De Klerk, 2004; Smit, 2004; Ward, 2005; Sivakumar, 2007; Wigley *et al*., 2009; Dube *et al*., 2011).

Globally, shrub thickening has become a threat to many savanna ecosystems with major economic and ecological impacts (Archer *et al*., 1988; Dougill *et al*., 1999; Silva *et al*., 2001; Asner *et al*., 2003; Fensham *et al*., 2005) including southern Africa (Trollope, 1980; Hoffman & Ashwell, 2001; Meik *et al*., 2002; Moleele *et al*., 2002; De Klerk, 2004; Smit, 2004; Joubert *et al*., 2008). Shrub thickening is a serious problem affecting both livestock and game ranching areas. It also commonly occurs in arid to semi-arid regions where there is not enough fuel loads from the herbaceous layer for fires to control the thickening of shrubs (Ward, 2005). This is particularly true for the southern water-limited Kalahari semi-arid rangelands, where serious thickening by woody increaser species such as *Acacia mellifera, Dichrostachys cinerea* and

Terminalia sericea contributes to the low occurrence and production of herbaceous plants (Smit, 2004; Scholes, 2009; Ward & Esler, 2011; Kgosikoma *et al*., 2012).

Shrub thickening was first recorded in South Africa in the 1920's and 1930's in the savanna regions of the Northern Province and the KwaZulu-Natal Province and was also later recorded in the 1940's in the savanna regions of the Kalahari (Hoffman & Ashwell, 2001). It is regarded as part of the process of rangeland degradation, since the increase in woody plant density results in a decrease in palatable, climax grass species. A workshop was held in 1980 where the South African Department of Agriculture, together with scientists, estimated the extent of the shrub thickening problem in South Africa. Of the 38 million ha of rangelands that were analysed, it was found that 4% (i.e. 1.52 million ha) was heavily encroached, 24% (9.12 million ha) lightly to moderately encroached, 19% (7.22 million ha) vulnerable to bush thickening and the remaining 54% was unaffected (Hoffman & Ashwell, 2001). After evaluating the problem of shrub thickening more recently in the North-West, Northern Cape, Eastern Cape and Limpopo Provinces of South Africa, the results showed that 42% of the rangelands within these respective provinces were already influenced by shrub thickening (Hoffman & Ashwell, 2001). In fact, according to Ward (2005), 10 to 20 million ha of rangelands in South Africa have seen a decrease in grazing capacity and biodiversity due to shrub thickening.

The success of woody plant establishment is mostly affected by competition for soil moisture with other woody or grass plants (Smith & Grant, 1986; Grundy *et al*., 1994; Jurena & Archer, 2003). The competition is asymmetric: Mature woody plants are competitively superior to grass plants, while grass plants are able to outcompete immature woody plants (Ward, 2005; Scholes, 2009). Accordingly, seedling survival of common savanna species such as *A. mellifera* and *A. erioloba* can be reduced significantly under the canopy of woody plants compared to further away from the canopy (Barnes, 2001; Joubert *et al.,* 2013).

One of the most serious woody increaser species is *A. mellifera.* It produces quite large seeds compared to other *Acacia* species such as *A. karroo* and *A. nilotica.* For this reason, *A. mellifera* requires greater resource investment to produce larger seeds (Joubert *et al.,* 2013) and, therefore, exerts greater competition with other plants for available moisture. Although mechanical or chemical mitigation practices

implemented to reduce shrub thickening can result in immediate changes in the competition between woody plants, no dormancy mechanism is observed in the seeds of *A. mellifera*, and only an adequate amount of precipitation is required for the seed to germinate successfully (Joubert *et al*., 2013). New individuals can then establish in the open spaces created by woody plant thinning actions (Smit, 2004).

The interrelated effect of both biotic and abiotic drivers of shrub thickening may inhibit vegetative growth and reproduction of woody and grass plants. Direct and indirect anthropogenic influences have modified the drivers of savanna systems. Drivers are either primary (e.g. climate and soil characteristics) or secondary (e.g. fire and herbivory). High densities of browsing ungulate species in the past were capable of significantly modifying the plant structure and composition in African savannas by utilising mainly woody plants. The removal of browsing ungulate species for commercial livestock farming practices may have attributed to the shrub thickening problem (Smit, 2004). The latter can be imposed by the primary global drivers, such as climate change and annual precipitation (Condon, 1986; Schlesinger *et al*., 1990; Fensham *et al*., 2005; Joubert *et al*., 2008), atmospheric nitrogen composition (Brown & Archer, 1987), elevated carbon dioxide (Polley, 1997; Bond & Midgley, 2000; Hoffman & Ashwell, 2001; Bond *et al*., 2003[b]) and soil characteristics (Mourik *et al*., 2007; Sankaran *et al*., 2008). In addition, plant structure and composition can often be modified directly by rangeland management systems (Smit, 2004) such as the exclusion of grazing and fire (Silva *et al*., 2001; Van Langevelde *et al*., 2003; Condon & Putz, 2007; Müller *et al*., 2007), incorrect burning practices (Trollope, 1992; Higgins *et al*., 2000; Bowman *et al*., 2001; Briggs *et al*., 2005; Brook & Bowman, 2006), changes in land use with the replacement of most indigenous browsers and grazers by domestic livestock species, often at high stocking rates (Huntley & Walker, 1982; Van Vegten, 1983; Archer *et al*., 1995; Roques *et al*., 2001; Van Langeveld *et al*., 2003; Sharpe & Bowman, 2004; Fensham *et al*., 2005; Wigley *et al*., 2010), poor grazing management practices and the instalment of artificial watering points (Smit & Retham, 1999).

The effect of mismanagement during severe droughts is much greater on the grass layer than on the woody layer. Climate change results in an increase in the evaporation rates and a reduction in the precipitation rate in dryland regions (Safriel, 2009; Archer & Tadross, 2009). Warm and drier climates may favour the woody layer

as this layer has a better adapted root system to survive periods of severe water stress. The general limiting resource of primary productivity in drylands is the fixing of atmospheric carbon through photosynthesis, which will be impaired by lower soil moisture contents (Safriel, 2009; Cowie *et al*., 2011). As a result, biological productivity is expected to decrease within drylands (Safriel, 2009).

The amount and frequency of precipitation play a vital role in the occurrence of shrub thickening. More precipitation is essential for woody plants to be able to germinate, compared to grass species (Smit, 2004). According to Joubert *et al*. (2013), at least two consecutive, proper rainfall seasons are required to ensure the successful recruitment of woody plants. Woody species may also germinate in larger numbers and are able to survive during very high rainfall years, be it with or without grazing (Scanlan & Archer, 1991; O'Connor & Crow, 1999; Smit, 2004; Ward, 2005).

To survive long drought periods in semi-arid regions, woody plants have developed vast root systems capable of utilising deep sources of moisture (Ingrouille & Eddie, 2006; Ward *et al*., 2012). Ward (2005) and Joubert *et al.* (2013) also documented the seed production and germination of *A. mellifera* as being positively related to precipitation. However, fire and grazing practices did not have an effect on tree seedling germination and the survival of *A. mellifera* in field and pot experiments conducted by Ward (2005).

Fire as a determinant of shrub thickening has been widely studied (Trollope, 1980, 1992; Trollope & Tainton, 1986; Higgins *et al*., 2000; Bowman *et al*., 2001; Briggs *et al*., 2002, 2005; Trollope *et al*., 2002; Bond *et al*., 2003 [a, b]; Brook & Bowman, 2006). Fire would act as the principal disturbance that prevents woody plants from dominating the grass plants (Scholes, 1997). However, studies have shown that fire alone is not effective enough to prevent shrub thickening in the savannas of southern Africa (Trollope & Tainton, 1986; Trollope *et al*., 2002). The total exclusion of fire or the mismanagement of burning practices on rangelands could, however, result in the increase of woody plant densities (Scholes, 1997; Smit, 2004).

Moleele and Perkins (1998) as well as Skarpe (1990[a]) state that cattle densities in savanna shrubland ecosystems are possibly the main driver for variation in the thickening by woody species around watering points at the expense of the grass cover and composition. Wigley *et al*. (2010), however, studied the thickening of

woody plants in savanna ecosystems under three very different land tenure systems (conservation, commercial and communal rangelands). Despite the differences in land-use systems and cattle densities, there was a significant increase in woody plant cover in all three systems from the year 1937 to 2004. In a study conducted by Smit and Rethman (1992), it was also reported that there was an increase in the woody plant densities over a 52-year period regardless of the grazing treatment implemented. However, the rate of thickening is especially faster and more intense in areas that had been severely grazed during the growing season.

According to Ward *et al*. (2012), most literature ascribes the co-existence between the woody and grass layers to spatial resource partitioning (e.g. Walter, 1939; Walker & Noy-Meir, 1982; Ward, 2005). This means that the roots of woody and grass plants occupy different vertical niches, where each vegetation stratum has exclusive access to water and nutrient resources in the soil (Ward *et al*., 2012). In other words, if woody and grass plants niches do not overlap, the plants seemingly would not have to compete for resources and, therefore, the woody-grass co-dominance would be in a stable equilibrium (Ward *et al*., 2012).

There are two models that discuss possible interactions between woody and grass plants. To explain root niche separation, Walter's two-layer model (Walter, 1939) is mostly used (Ward *et al*., 2012). This model states that the root system of both woody and grass plants are located at the top soil layer. However, the roots of the woody plants also have access to deeper sub-soil layers. This hypothesis then suggests the roots of woody and grass plants do not have exclusive access to moisture in the top-soil layers, resulting in competition for moisture and nutrients between the woody and grass layer (De Klerk, 2004; Ward & Esler, 2011).

In areas with less than 250 mm MAP, grasses would be the superior competitor because all the moisture would be readily absorbed by the intensively transpiring and fast growing grasses with their dense shallow root systems. Under such dry conditions, woody plant seedlings would not be able to establish due to the great water-use efficiency of grasses (Ward, 2005; Ward *et al*., 2012). Above 250 mm MAP, not all the water can be absorbed by the grasses, and the water penetrating deeper into the sub-soil is exclusively utilised by the woody plants. Greater access to moisture in the sub-soil layers benefits the woody plants, resulting in these plants

becoming the superior competitor within the vegetation layer (De Klerk, 2004; Ward, 2005; Ward *et al*., 2012). Superior competitive encroaching and thickening woody plants suppress the grass layer, creating bare open soil patches that allow woody seedlings exclusive access to moisture in the top soil to recruit *en masse* (De Klerk, 2004; Ward, 2005; Ward *et al*., 2012).

Although Walter's two-layer model (Walter, 1939) implies that the major limiting factor is moisture, key nutrients in the soil, such as nitrogen (N) and phosphorus (P), are also limiting factors (Davis *et al*., 1999; Dougill *et al*., 1999; Kraaij & Ward, 2006). Many shrub-encroached areas are or have been subjected to heavy grazing, but this does not mean overutilisation is the cause of shrub thickening or that Walter's model is necessarily correct (Wiegand *et al.,* 2006). Some studies have completely rejected this theory (e.g. Brown & Archer, 1999; Higgins *et al*., 2000; Ward, 2005; Wiegand *et al*., 2005). It is predicted by Ward *et al.* (2012) that within arid savannas, the hypothesis of Walter's two-layer model would be able to predict root-niche partitioning, but this would not be possible within moist savanna systems because of the other factors that play a larger role in these systems, such as herbivory, fire intensity and physical soil disturbances (Ward *et al.,* 2012).

Shrub thickening has also been observed to occur on a single soil layer that does not allow for root separation between woody and grass plants and where grazing is light and infrequent (Ward, 2005; Wiegand *et al*., 2005; Ward, 2010), thus rejecting Walter's two-layer model hypothesis. Wiegand *et al.* (2005) hypothesised that within ecological systems of many semi-arid regions, the thickening of shrubs is a natural phenomenon governed by patch-dynamic processes. Any form of a disturbance, be it grazing, fire or annual drought, is capable of creating open spaces that make moisture and nutrients freely available for woody plant germination (Ward, 2005).

Precipitation is also patchily distributed over time and space in most savanna systems and, therefore, it will be a very rare occurrence for a spatial overlap of unusual high frequency rainfall events within the same year (McNaughton, 1985; Ellis & Swift, 1988; Ward *et al*., 2004; Ward, 2005). The patchiness of precipitation will lead to the overall patchy distribution of vegetation patterns, sometimes only a few hectares in size (Ward, 2005). With a low annual average precipitation, tree

growth is prohibited by insufficient soil moisture, while at or above a certain precipitation level, dense stands of woody plants develop (Ward, 2005).

Intensive grazing depletes grasses to such a state that they can no longer utilise the moisture and nutrients from the top soil layer effectively. This effectively weakens the suppressive effect grasses have over young woody plants which, in turn, increase in abundance and can transform open savannas into woodlands. The mean distance between neighbouring woody plants increases as the size of the woody plant increases. This is true for many savanna plants, especially the genus *Acacia* which are very 'canopy intolerant' (Wiegand *et al*., 2006). Weaker plants and seedlings are eliminated by stronger neighbouring woody plants; therefore, the plants become more evenly spaced (Ward, 2005; Wiegand *et al*., 2006). Consequently, the continuous growth and recruitment of woody plants in transformed woodlands can lead to inter-woody plant competition in future which can, ultimately, convert the shrub-encroached area back to an open savanna system (Ward, 2005).

Taking all of this into account, the idea that heavy grazing by domestic livestock and the exclusion of fire are the sole reasons for shrub thickening could be wrong (Ward, 2005; Wiegand *et al*., 2005). There are more influencing factors that play a role in this process of woody plant dominance over the grass layer. On a landscape scale, savanna systems can be stable as the system is composed of various patches that are in different states of transitions between grass and woody plant dominance (Ward, 2005). This concept therefore claims that shrub thickening is an essential part of savanna dynamics (Ward, 2005).

2.4 Restoration of degraded savanna ecosystems

Ecological restoration is implemented by restoration ecologists and land users to accelerate and assist with the recovery of a degraded, damaged or destroyed ecosystem. Ecological restoration aims to restore a resilient, self-sustaining ecosystem and to recover lost ecosystem goods and services while simultaneously enhancing the livelihoods of so many living in degraded drylands (Hobbs & Norton, 1996; Aronson *et al*., 2007; Bullock *et al*., 2007; 2011; Kellner, 2008; Cortina *et al.,* 2011). Consequently, land users and managers of degraded rangelands must implement restoration practices to improve such lands' productive capability (Hobbs

& Norton, 1996; Chirino *et al*., 2009) whilst setting quantitative goals and employing an unambiguous assessment methodology (Kellner, 2008).

Restoration of degraded rangelands is not only restricted to the environmental conditions of the rangeland but also has to do with active intervention as far as social and economic conditions are concerned (Van Andel & Aronson, 2006; Kellner, 2008). The aim of restoration is to retain a system with its two main components – ecosystem function (forage production and soil nutrient content) and ecosystem structure (species composition and complexity) – since even after the disturbance may have ceased, degradation could still continue, or remain stable or even recover at various rates. The relationship between rangeland functionality and structure will be determined by the needs and objectives of the land user (Van Rooyen, 2000).

Natural successional processes (i.e. from pioneer to climax grass species) in semi-arid savannas require longer time periods to take place, making restoration practices and better rangeland management practices a necessity in order to accelerate the grass plant succession process (Bradshaw, 1994 *in* Van Rooyen, 2000; Snyman, 2003). Rangeland restoration is a long-term process requiring considerable pre-planning and financial inputs to restore sustainable livestock production (Dregne, 1995; Snyman, 2003; Van Andel & Aronson, 2006; Kellner, 2008). Restoration practices need to take into account the climatic variability which will result in delayed vegetation responses to management (Curtin, 2002).

During the pre-planning stage of a restoration process, a reference ecosystem and/or ecosystem function (e.g. increase grass forage production) that is worth restoring need to be identified (Aronson *et al*., 1993; Kellner, 2008; Chirino *et al.,* 2009*)*. The reference system or function thus identified will be the objective the land user wants to achieve once restoration has been completed. According to Kellner (2008) and Chirino *et al.* (2009), dryland ecosystems are dynamic and, therefore, the temporal and spatial scale requires careful consideration when deciding on and identifying the reference ecosystem and type of restoration practice to achieve the objective. Restoration practices implemented to restore the reference ecosystem should be based on sustainable ecological principles and processes to ensure the successful restoration of degraded rangelands (Visser *et al*., 2004; 2007). Restoration success in savanna drylands will largely be influenced by climatic

conditions, restoration costs, the nature and extent of degradation and the post-management practices applied to restored rangelands (Snyman, 2003).

Passive restoration interventions in semi-arid savannas are applied in systems with a high resilience and limited functional damage (Visser *et al*., 2007; Kellner, 2008). These interventions include removing stresses such as heavy grazing by implementing rotational grazing management (withdrawing livestock) to allow vegetation with longer periods of rest to recover (Whisenant, 1995; Milton & Dean, 1996; Snyman, 1998; Tainton *et al*., 1999; Curtin, 2002; Müller *et al*., 2007; Scholes, 2009). In instances of a limited occurrence of rangeland degradation, these systems are capable of self-recovery due to the available seed remaining in the soil-seed bank (Kellner, 2008). If the land user has the knowledge to implement sustainable rangeland management practices, low input costs would be required and the practices can be affected with relative ease. The impacts from the hooves of livestock on the soil surface and uniform utilisation of vegetation are regarded as positive tools for rangeland restoration (Briske *et al*., 2011). If the plants are under-utilised and fewer disturbances occur, a decline of soil conditions could result which may lead to competition between plant species (Briske *et al*., 2011).

Active restoration interventions are applied in savanna systems that lost their function and structure with low-resilience, high-encroacher woody plant densities, low vegetation cover and density (Aronson *et al*., 1993; Kellner, 2008). To facilitate better grass establishment and reduce woody densities, active restoration interventions will include shrub clearing (mechanical or chemical), veld burning, brush packing, fertilisation and/or cultivation (Milton & Dean, 1996; Visser *et al*., 2004; 2007; Scholes, 2009; Teague *et al*., 2010). The soil seed bank in these degraded rangelands is usually depleted and will require re-vegetation through reseeding practices to ensure re-establishment of perennial plant species (Milton, 1994; Van den Berg & Kellner, 2005; Kellner, 2008; Scholes, 2009).

Certain South African savanna rangelands have been degraded to such an extent where the implementation of improved management practices and/or even the complete withdrawal of livestock would not be able to ensure recovery of rangeland production whatsoever (Snyman, 2003). Therefore, the type and degree of degradation, soil type and resources available will determine the type of practices to

be applied (Snyman, 2003; Kellner, 2008). Local land users are mostly interested in restoration practices that are inexpensive and would yield the best results over a very short period of time. Land users are also very sceptical about rangeland restoration as it contains an element of risk (Milton & Dean, 1996; Snyman, 2003).

The land user does not know whether the rangeland would receive enough rainfall after a lot of money and time have been spent on re-vegetating a large area, or even whether a fire after a year of re-vegetation will be devastating. In most cases, local land users do not have the financial capacity or time to leave grazed land for longer recovery periods. Consequently, they will rather opt to implement active restoration interventions (Van den Berg., 2007). However, it is still necessary to carry out restoration alongside other rangeland management practices, such as decreasing the stocking rate on the restored land and applying resting periods. This would give the vegetation time to re-establish and recover, ensuring better restoration success in savanna ecosystems (Van den Berg, 2007).

In the section to follow, special attention will be paid to the restoration practices implemented in southern African degraded savanna rangelands, including the semi-arid parts of the Kalahari addressed in the present study.

If woody plant densities are high and increased forage production is desired, woody plant thinning will be required to attain a predetermined density, after which a post-thinning management programme needs to be implemented to sustain an open grassy area for livestock production (Smit, 1994). The cutting of woody plants is a common practice to control shrub thickening (Smit, 2003[a]). Woody plants have a strong re-growth ability following mechanical shrub control measures (Teague & Smit, 1992; Smit & Rethman, 1998; Richter *et al*., 2001; Smit, 1994, 2003[a]). To prevent woody plant re-growth, it is suggested that the stumps of woody plants that were cut be treated with arboricides followed up by yet another mechanical cutting or to make use of fire in future if re-growth occurs (Smit *et al*., 1999; Richter *et al*., 2001; Smit, 2004). Through the removal of unwanted woody plants, the increase in grass forage production can be stimulated, thereby increasing the grazing capacity of the rangeland (Smit *et al*., 1999; Smit, 2004). The effectiveness of woody plant control actions would be determined by factors such as MAP and the vegetation type.

Relatively effective arboricides are commercially available in South Africa to control unwanted encroacher woody plants, a practice already well-established in many parts of the Kalahari bushveld (Hagos & Smit, 2005). However, chemical shrub control needs to be considered carefully before applying as a method to restore the agricultural potential of rangelands. According to Smit *et al*. (1999), aspects to consider before applying chemicals as a restoration practice include: (a) whether sufficient available grass material (fuel) exists to, perhaps, support a fire that will be sufficient enough to kill the top-growth of the encroacher woody species; (b) whether the majority of the trees have grown to a height beyond the reach of browsing animals, in which case arboricides may be required; (c) whether the woody layer has become so dense that animal movement is restricted; (d) whether the woody plants are largely unpalatable; and (e) whether, for some reason, it is not practical to incorporate browser species together with the grazing livestock in the system.

Factors affecting the dosage and effectiveness of the chemical are the clay content of the soil (the higher the clay content, the greater the dose), organic matter content and pH of the soil, the species of woody plant treated and the size thereof (Moore *et al*., 1985; Smit *et al*., 1999; Dube *et al*., 2011). The season during which the chemical treatment is applied is also of considerable importance. When the carbohydrate reserves in the woody plants are low, the plants will be more susceptible to external interference (De Klerk, 2004).

According to Smit (2004), chemical control will need to comply with two important requirements: The control should be ecologically sustainable and also economically justifiable. It is very costly to implement chemical control, and the correct approach has to be followed, as re-thickening by woody plants can and most probably will follow (Smit, 2004). It should, nevertheless, be kept in mind that woody plants have an ecological importance in savanna ecosystems (Kelly, 1977; Donaldson, 1979; Fagg & Stewart, 1994; Smit, 2004).

Failures of re-vegetation with indigenous species in semi-arid savannas are common due to the variable annual precipitation (Snyman, 2003). Even after re-vegetation, degraded areas may still be unstable and be dominated by mostly pioneer species. Climax grass species such as *Anthephora pubescens* are not able to establish very rapidly on bare soil patches where the formation of crust has already occurred.

Therefore, applying progressive soil cultivation methods before seed sowing can ensure the best survival rate and will encourage the infiltration of water into the soil (Rietkerk & Van de Koppel, 1997; Montalvo *et al*., 2002; Snyman, 2003; Kellner, 2008, Van Oudtshoorn, 2012). This will ensure that the soil seed bank is continually improved (Snyman, 2003) and infiltration reduction on sandy soils does not occur (Scholes, 2009). Local ecotype grass species should be used in re-vegetation practices as these species are better adapted to the soil and climatic conditions (Kellner, 2008; Van Oudtshoorn, 2012).

It is recommended that the re-vegetated areas be left un-grazed for at least a year after good precipitation and, during the second to third year, the re-vegetated area be grazed at a light intensity only (Van der Merwe & Kellner, 1999). Follow-up management that is applied to the restored rangeland will ultimately determine the success achieved with re-vegetation (Snyman, 2003). Management should take into consideration and mainly focus on the new grass species introduced into the system to afford these newly seeded grass species the opportunity to produce seed (Snyman, 1999).

According to Van Oudtshoorn (2012), the following important factors need to be considered to ensure successful re-vegetation practices: (a) season selection for sowing in seeds, (b) seeding depth and density, (c) clearing area of woody species and (d) the fertilisation of the soil. A study conducted by Snyman (2003) on various re-vegetation practices on semi-arid rangelands of both sandy and clayish soil types highlighted that two grass species, *Anthephora pubescens* (Bottlebrush grass) and *Cenchrus ciliaris* (Blue buffalo grass), are highly capable of establishing successfully in sandy soils. These are also the two grass species used in the Molopo rangelands for re-vegetation practices.

Rangeland restoration (e.g. chemical shrub control and re-vegetation) is a long-term process that would not be able to deliver sustainable livestock production over the short term (Snyman, 2003). According to Sankaran and Anderson (2009) and Higgins *et al*. (2007), economic models of rangelands suggest that a sustainable long-term strategy to manage and restore savanna ecosystems conservatively will include a low-level stocking rate and controlling the tree-grass ratio – be it by means

of fire or chemical/mechanical control. In this instance, prevention, therefore, is better than cure (Sankaran & Anderson, 2009).

To maximise restoration success, it has been suggested by Holmgren and Scheffer (2001) that managers should implement restoration practices, e.g. re-vegetation, during the El Niño Southern Oscillation in order to make use of the anticipated high precipitation. To enhance the restoration process further, more social involvement by the local community and land users is required (Stringer & Reed, 2007; Cortina *et al*., 2011). If the people affected by rangeland degradation are not involved in the process, the opportunity to develop adaptive management and improve the effectiveness of different restoration and management practices will be missed (Badejo, 1998; Cortina *et al*., 2011). According to Curtin (2002), Kellner (2008) and Cortina *et al*. (2011), prescribing more efficient restoration practices hinges on gaining more insight into the ecology of key species as well as the ability to monitor restoration success over the long term. It is further claimed by Cortina *et al*. (2011) that unless integrated and participative management strategies and research are implemented, mitigation practices would have less of an impact on restoring ecosystem goods and services and improving the livelihoods of people in dryland systems.

2.5 Recent developments in participatory research in drylands

Combating rangeland degradation is a complex process and, consequently, requires integrated and multidisciplinary research, encompassing local land users since they are the main implementers of rangeland management practices (Von Maltitz, 2009). In South Africa, a need exists for an information base that will assist land users in attaining sustainable rangeland management, especially in rural areas still subject to inappropriate policy frameworks (Von Maltitz, 2009; Nkonya *et al*., 2011; Vetter, 2013). This can be best achieved through participatory research: An integrated process that combines local indigenous knowledge with scientific expertise and actively involves affected land users in evaluation, decision-making and execution processes (Fraser *et al*., 2006; Reed *et al*., 2006; Briske *et al*., 2011).

Indigenous knowledge is broadly defined by the United Nations Environment Programme (UNEP, 2009) as the rich set of experiences and possible explanations for changing environments that an indigenous (local) community or individual land user develops over many generations of living in a particular environment. The definition encompasses all forms of knowledge, technologies, know-how, skills, practices and beliefs that enable the individual land user to achieve stable livelihoods within their environment (UNEP, 2009).

An integration of this indigenous knowledge with ecological scientific knowledge to combat rangeland degradation (i.e. focusing on the rangeland management practices being implemented by the land users in semi-arid rangelands) would yield more information on the dynamics of ecosystems and rangelands and the response to management than sole use of ecological methods would (Sheuyange *et al*., 2005; Fabricius *et al*., 2006; Reed *et al*., 2007; Stringer & Reed, 2007; Verlinden & Kruger, 2007; Oba *et al*., 2008[b]). Local communities are rarely involved in scientific top-down approaches and are almost never privy to any feedback from projects that could have insured more sustainable management or restoration of their degraded land (Reed *et al*., 2007; Briske *et al*., 2011). Science's top-down approaches towards rangeland degradation usually focus on single issues and, as a consequence, the multi-faceted nature of the problem contributing to rangeland degradation is never addressed (Reed *et al*., 2007).

During the late 1960's, a need for sustainable approaches to stakeholder participation first came to the fore (Reed, 2008). During the period 1970 to 1990, several attempts were made to incorporate local perspectives into the scientific data collection, evaluation and decision-making process (Reed, 2008), thereby acknowledging the ecological and restoration value local knowledge could add to those experiencing rangeland degradation (Dahl & Hjort, 1976; Western, 1982; Chambers, 1983, Chambers *et al*., 1989) and giving rise to participatory research.

Over the years, developments in this regard were confined to parallel geographical and also disciplinary contexts (Reed, 2008). In the process, lessons were learned from different disciplinary contexts such as social activism, adult education, natural resource management and ecology (Reed, 2008). However, in the course of applying participatory approaches in varying disciplinary contexts, this methodology

gained ideological, social, political and also methodological meaning, resulting in a wide array of interpretations (Lawrence, 2006; Reed, 2008; Stringer *et al*., 2009). Through the years, different typologies developed in an attempt to gain a better understanding of the differences between the various interpretations and also the associated approaches employed (Reed, 2008). Likewise, developing an array of typologies in an attempt to understand the varying contexts most suited to participatory research would be justified (Reed, 2008).

In order to determine which participatory method to implement with regards to the type of participation the typologies may be used a priori (Reed, 2008). However, to categorise which participation type occurred, the typologies may be used *post hoc* (Reed, 2008). In addition, Reed (2008) held that these typologies should only serve as alternative platforms from where ultimate stakeholder participation ought to be reached. As such, the platform will help to decide on the most appropriate method in a given context with the objective to attain a given goal (Reed, 2008).

Reed *et al*. (2007) examined the possibility of identifying suitable rangeland management practices by combining local and scientific knowledge. In this instance, the authors applied a four-stage approach to stakeholder participation in encouraging social learning. During the first stage, literature was used to identify current and possible sustainable management practices. Semi-structured interviews with the local land users were carried out during stage 2 to gain local knowledge regarding rangeland degradation within the region. The knowledge and expertise gained in stage 2 were then discussed during stage 3 of the process within focus groups comprising of various rangeland stakeholders from across the region. The fourth and final stage made use of the outputs gained from these focus groups to produce sustainable rangeland assessment and management actions to combat rangeland degradation within the region (Reed *et al*., 2007).

Other studies (see Oettlé *et al*., 2004; Malgas *et al*., 2010; Koelle, 2013) integrated local- and scientific knowledge by implementing a trans-disciplinary approached generally known as PAR (Participatory Action Research) (Kemmis & McTaggart, 1988). In their attempts to arrive at appropriate strategies (i.e. shrub-control methods and rotational grazing management) in order to gain a better understanding of how rangeland degradation and shrub thickening can be reduced and combated, Seely

(1998), Reed *et al*. (2007) and Stringer and Reed (2007) found the benefits and importance of incorporating local knowledge into the decision-making process to be invaluable. On the subject of improving restoration research, planning and management, Botha *et al*. (2008) pointed out that qualitative interviews conducted with local land users could well afford an opportunity to tap into first-hand knowledge gained through many years.

The quality and durability of decisions regarding restoration and management practices are likely to be greater when involving the local land users and communities (Beierle, 2002; Reed *et al*., 2007; Reed, 2008; Briske *et al*., 2011). Local land-user and community participation may also increase the public's trust in the proposed decisions if the participatory processes are transparent and consider conflicting claims and views (Reed, 2008; Briske *et al*., 2011). According to Reed (2008), stakeholders contribute to the decision-making process when, in cooperation with scientists, local knowledge is generated and furnished to ensure local participants can put findings thus generated to best possible use. Blackstock *et al*. (2007) and Fernandez-Gimenez *et al.* (2008) even hold that participatory research could be instrumental in promoting social learning. This does not necessarily lead to changes in attitudes, but individuals and the wider society come to understand and appreciate their own and others' interests, values and experiences in a social situation by developing relationships or building on existing relationships (Bosch *et al*., 1996; 1997; Stringer *et al*., 2006; Blackstock *et al*., 2007; Reed, 2008). In this way, diverse groups of potentially conflicting stakeholders may develop, through working together, more creative solutions resulting in higher quality decision making to combat rangeland degradation after long and careful considerations (Beierle, 2002; Stringer *et al*., 2006; Reed *et al*., 2007). This can create a sense of ownership for the land users over the participatory process and the outcomes thereof (Reed, 2008). Participation enables mitigation practices to be better focussed and adapted to the socio-cultural and environmental conditions of the local community (Botha *et al*., 2008; Reed, 2008). This would ultimately ensure long-term support and more willingness by the land users to adopt these actions as it then considers their needs and priorities (Reed, 2008).

Reed (2005) claims that there is a need to aim towards local sustainability when trying to address rangeland degradation, and he defines local sustainability as

developing the economy and environment of the local community. This process to combat degradation sustainably should be led by local members of the community that will address issues regarding rangeland degradation whilst simultaneously maintaining the resources required to ensure a better quality of life for both present and future generations (Reed, 2005). The integration of biophysical as well as socio-economic information gained from a wide range of different sources and scales is growing in importance towards effective environmental sustainability assessments (UNCCD, 1994; Land Degradation in Drylands, LADA, 2001; Warren, 2002; Reed, 2005).

According to Reed (2008), there is, however, concerns about stakeholder participation not taking place because of the power vacuum within existing power structures where groups previously marginalised now exert negative attitudes towards power structures. Group dynamics within the participatory approach may discourage the expression of the minority's perspectives. Stakeholders are also continually asked to participate in workshops, resulting in the possibility of consultation fatigue, especially if the workshops are not well organised and if the stakeholders perceive that their involvement in the project will yield no reward and will have a negligible impact on the outcome (Reed, 2008). Then the question arises whether stakeholders have sufficient expertise to contribute to debates which are often of a highly technical nature. Local indigenous knowledge still needs to be verified by quantitative scientific research to ensure that valuable information is provided to the local land users (Reed *et al*., 2006; Reed *et al*., 2007; Oba *et al*., 2008[b]) and to prevent decisions being made regarding rangeland management based on unverified local assumptions (e.g. Sullivan, 2000).

Only a limited number of investigations as to the validity of stakeholder participation has been conducted (Blackstock *et al*., 2007; Reed, 2008) since the focus seems to be on evaluating the participatory process and not judging the outcome thereof (e.g. Rowe & Frewer, 2000; Beierle, 2002). The methodology used in participatory research to determine and gain this knowledge also needs to be verified carefully as conceptual thinking and political factors do play a definite role regarding the indicators (such as support from government, animal condition or personal gain) used to evaluate the effectiveness of different management actions (Reed *et al*., 2007; Cooke & Kothari, 2001).

2.5.1 Participatory indicator development

An indicator is defined as a relevant environmental component or measure to evaluate and describe changes in environmental conditions or preset environmental goals (OECD, 2003; Van Andel & Grootjans, 2006; Heink & Kowarik, 2010). The process of indicator identification and selection by stakeholders is to create a baseline of information so that the impact of mitigation practices implemented in rangelands can be evaluated. It also provides interdisciplinary information about the changes that take place within these environmental systems over certain time periods (Reed, 2005; Fraser *et al*., 2006).

To detect these environmental changes, effective assessments of these indicators within the rangeland management system need to take place. It is also evident that if the land users do not gain benefits from previously established sustainability indicators, they would, as a consequence, not make use of these indicators (Innes & Booher, 2000). This can partly be ascribed to one of two factors: (1) The indicators were identified by means of a top-down approach and the local land users were not included in the discussion and identification of these indicators, the latter mostly being developed by researchers, or (2) the indicators were derived without involving any experts (Innes & Booher, 2000; Reed, 2005). It should be ensured that the indicators selected take environmental aspects as well as the socio-economic aspects into consideration, and the indicators should then be linked to the needs, priorities and economic goals of the local land user. The importance of participation by local land users in selecting and monitoring evaluation indicators can, therefore, not be disregarded (UNCCD, 1994; Bossel, 2001; Riley, 2001; Reed *et al*., 2006). Regarding the monitoring of these indicators, the methods chosen should not be a complicated procedure. Rather, it should be a process local land users can administer on their own with little or no assistance.

Indicators can enhance the understanding of environmental, social and economic problems. It would empower the local land users to be able to respond appropriately in the face of environmental changes in a sustainable manner to combat rangeland degradation, without constantly having to rely on the knowledge of external experts (Reed, 2005). On a national and international scale, such an understanding involves studies on environmental change and sustainable development together with the

monitoring of each component to determine clear patterns of change in relation to the reference points/ indicators which can be used to influence policy change (Riley, 2001).

It can be concluded that rangeland sciences do play a key role in providing solutions to combat rangeland degradation and assessing the effectiveness thereof, but there are, however, limitations to the available scientific knowledge. It is not always possible to provide an accurate diagnosis or satisfactory solution to urgent problems, as restoration of degraded rangelands operates over longer timescales (Thomas, 1997). One form of knowledge should not be favoured above another, as this participatory research approach should lead to an integration of scientific and local knowledge for a better understanding of sustainability and the causes of vegetation change on rangelands within semi-arid regions (Thomas & Twyman, 2004; Briske *et al.*, 2011). This integration of scientific and local knowledge can assist in the development of more effective mitigation practices to combat degradation (Fraser *et al*., 2006; Reed *et al*., 2007; Stringer & Reed, 2007). Regrettably, although according to Stringer and Reed (2007) many studies are advocating this approach, very few are implementing it.

3

Study Area

3.1 Location and land use

The study area is situated in north-central South Africa within the Bophirima (Dr. Ruth Segomotsi Mompathi) agricultural region of the North-West Province approximately 100 km west of Mafikeng and 100 km north of Vryburg. It covers an area of around 12 588 km^2 (Figure 3.1). The Molopo local municipal area consists of rural service centres, communal rangelands, large commercial ranching and/or dry-land cropping farms (www.premier.nwpg.gov.za). The Molopo River stretches along the northern parts of the study area and forms the border between South Africa and Botswana. The local municipality is named after the Molopo River; consequently the study area will henceforth be referred to as the Molopo.

In the year 2011, the amalgamation process for the two local municipalities Kagisano and Molopo was completed to form a single local municipality called the Kagisano-Molopo local municipality. This amalgamation was proposed to enhance the institution's viability and to improve the provision of services provided by the government to the communities (www.info.gov.za).

The district's major rural land uses are comprised of small mines, tourist attraction facilities, isolated industrial activities, intensive agricultural activities and game farms for both hunting and ecotourism purposes (www.premier.nwpg.gov.za). The Molopo ranching area consists of semi-arid *bushveld* (Donaldson & Kelk, 1970). *Bushveld* refers to a vegetation layer with an herbaceous understory and a tree/shrub overstory (Trollope *et al*., 1990), such as the local savanna vegetation. In this study, considered ranching areas include Bray, Tosca, Pomfret, Vergeleë, Vostershoop, Gemsbokvlakte and Makopong.

The livestock in the Molopo consists predominantly of cattle, goats and sheep. Cattle and goat industries are largely based on savanna lands (Jacobs, 2000). Livestock

located on large, commercial rangelands are mainly for meat production. Meat production is a secondary objective on communally grazed lands, where livestock offer other products such as milk and is also seen as an accumulation of assets (Scholes & Archer, 1997). Communal lands typically have higher stocking rates compared to commercial rangelands. There are limitations to the agricultural potential in the Molopo for crop production. Low precipitation has a great impact on land use, making cultivation of crops very risky as it only produces low yields (Jacobs, 2000). Tourism, especially eco-tourism, has the potential to contribute to the economy of the Molopo in the near future as game farms and safari lodges are on the increase. During the winter months, the game farms are used for hunting purposes, catering especially for visiting foreign hunters from European countries and the United States. In the off-season, the game farms are also used for eco-tourism purposes to ensure a continuous income (www.premier.nwpg.gov.za).

Sustainable economic use of the Molopo rangelands requires adequate adaptation to this type of semi-arid environment. The growing use of the environment as a grazing source for cattle and small livestock has one of the most significant influences on the Kalahari environment (Thomas & Shaw, 1991; Jörg *et al*., 2004; Jörg *et al*., 2006). This development is possible due to the exploitation of ground-water aquifers through borehole construction, providing year-round water for both human and livestock use (Thomas & Shaw, 1991).

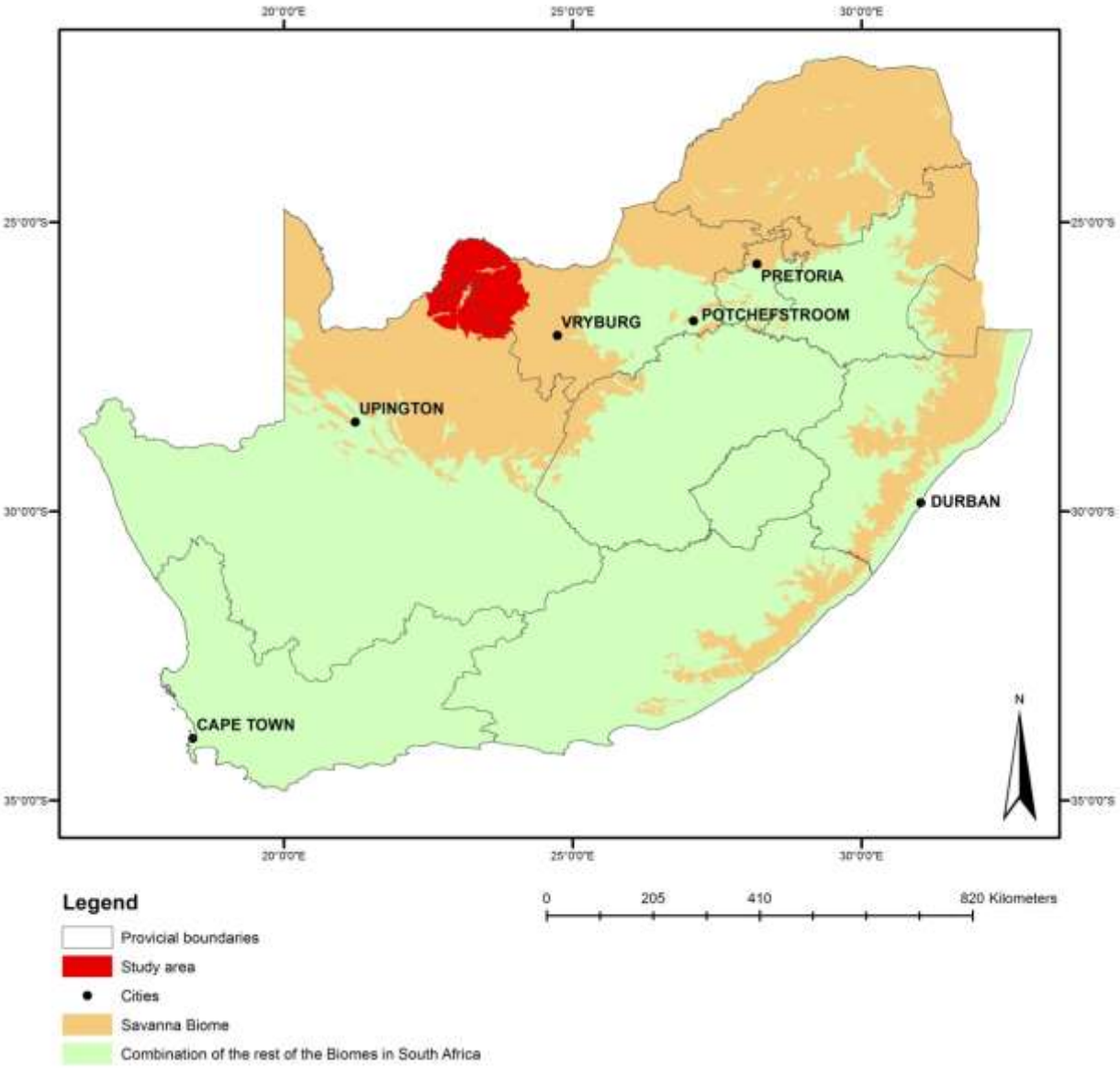

Figure 3.1: The location of the Molopo study area (Molopo Bushveld vegetation type) in the North-West and Northern Cape Provinces of South Africa within the Savanna Biome.

3.2 Climate

The Molopo experiences a predominantly semi-arid climate (Nash & Endfield, 2002; Van Niekerk, 2011), receiving rainfall during summer and early autumn. The precipitation levels are spatially and temporally highly variable (Nash & Endfield, 2002; South African Weather Service, 2013; Figure 3.2). Only climate data from two weather stations were available for the description of the climate for the Molopo. The first weather station, Severn [0428635 1], is located on the north-eastern boundary and the second weather station, Bray [0541297 5], on the south-western boundary of

the Molopo. The precipitation data over a time-span of 34 years was obtained from the South African Weather Services.

The coefficient of variation for the weather station Severn [0428635 1] is 41.8% and 47.4% for Bray [0541297 5] (evaluated from the annual rain data from 1980 to 2013 for the two weather stations within the Molopo region). The precipitation is the lowest in the southwest of the study area with a mean annual precipitation (MAP) of 250 mm and gradually increasing to 500 mm in the north-eastern regions of the study area (Low & Rebelo, 1998; Rutherford *et al*., 2006; South African Weather Service, 2013). The Molopo receives its highest monthly precipitation in the summer season (between November and April) and peaks in January (Figure 3.2 and Figure 3.3) (Tyson & Crimp, 1998).

Data from the weather station Severn [0428635 1] was used for the description of the precipitation on the south-western boundary of the study area. Average monthly precipitation for the years 1980 – 2013 is illustrated in Figure 3.2. January and February are the months when the area receives the highest average monthly precipitation, more than 47 mm each month. It is clear that the second part of the summer season receives the highest precipitation, with 74% of the annual rainfall occurring from November to March. Precipitation decreases profoundly at weather station Severn [0428635 1] during June to August, receiving less than 5 mm precipitation a month. The annual precipitation varied between 78.8 mm and 537.3 mm with an average of 273.4± 116.2 mm over the past 34 years (1980 – 2013) (Figure 3.3).

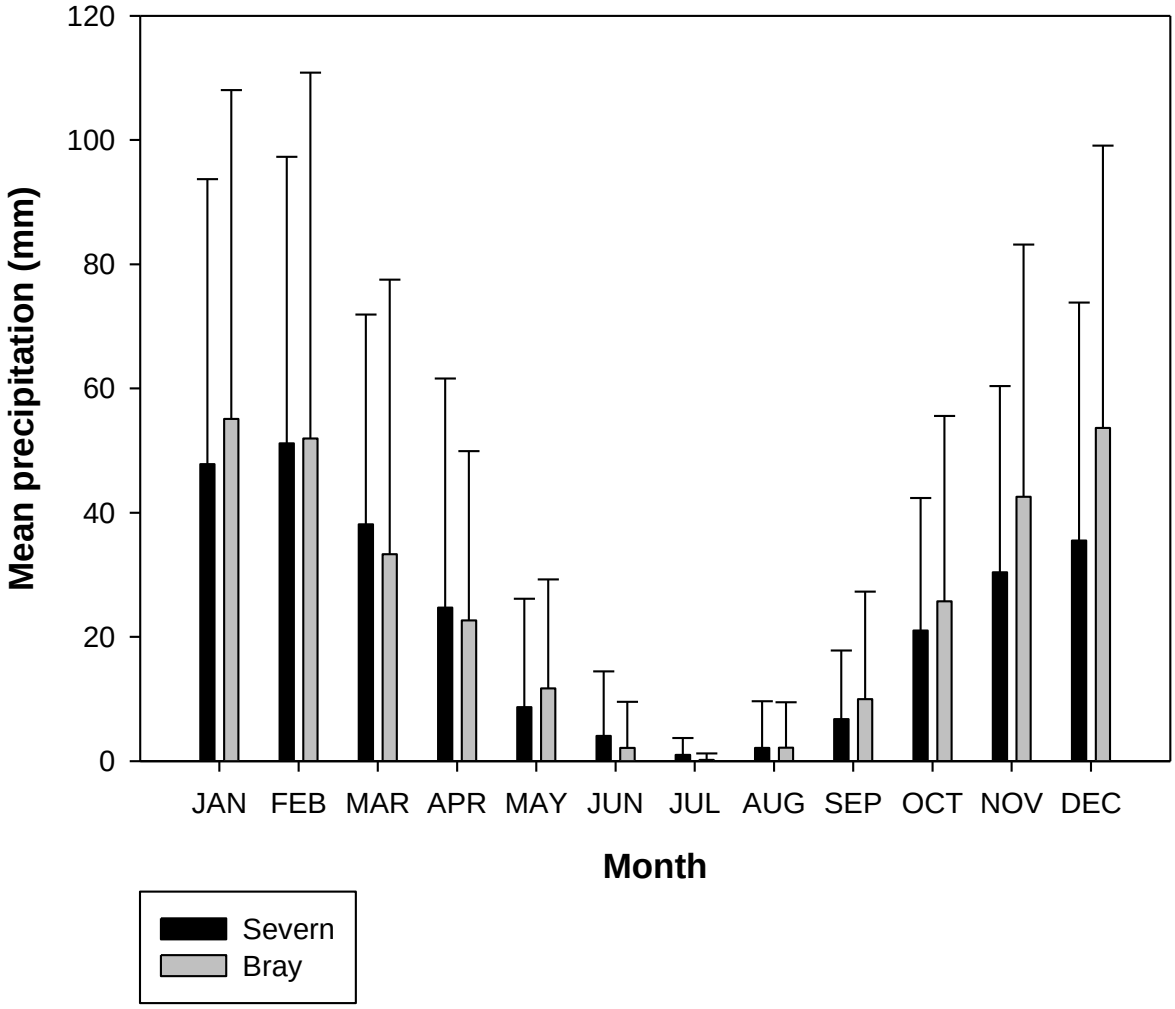

Figure 3.2: The mean monthly precipitation at the two weather stations Severn [0428635 1] and Bray [0541297 5] for the 34-year-period from 1980 – 2013 (data source South African Weather Services, 2013).

Data from the weather station Bray [0541297 5] was used for the description of the precipitation on the north-eastern boundary of the study area. Average monthly precipitation for the years 1980 – 2013 is illustrated in Figure 3.2. December up to February are the months when the area receives the highest average monthly precipitation, more than 50 mm each month, with 85% of the annual precipitation occurring from November to March. During June to August, the area receives a very low precipitation with less than 5 mm each month. The annual precipitation varies between 81.9 mm and 632.4 mm with an average of 314.7± 151.3 mm over the past 34 years (1980 – 2013) (Figure 3.3).

There are definite year-to-year variations in total annual precipitation (Figure 3.3) and seasonal precipitation patterns that contribute greatly to the character of the south-eastern Kalahari region as a semi-arid environment (Thomas & Shaw, 1991).

The mean annual temperature for the Molopo is 19.1^{o}C (Rutherford *et al*., 2006). Great seasonal and daily variations in temperature can be expected, varying between -9^{o}C and 42^{o}C (Mangold *et al*., 2002). Both mean daily maximum and minimum temperatures are higher during the wet- than during the dry season. Very dry winters and frequent frost can be expected.

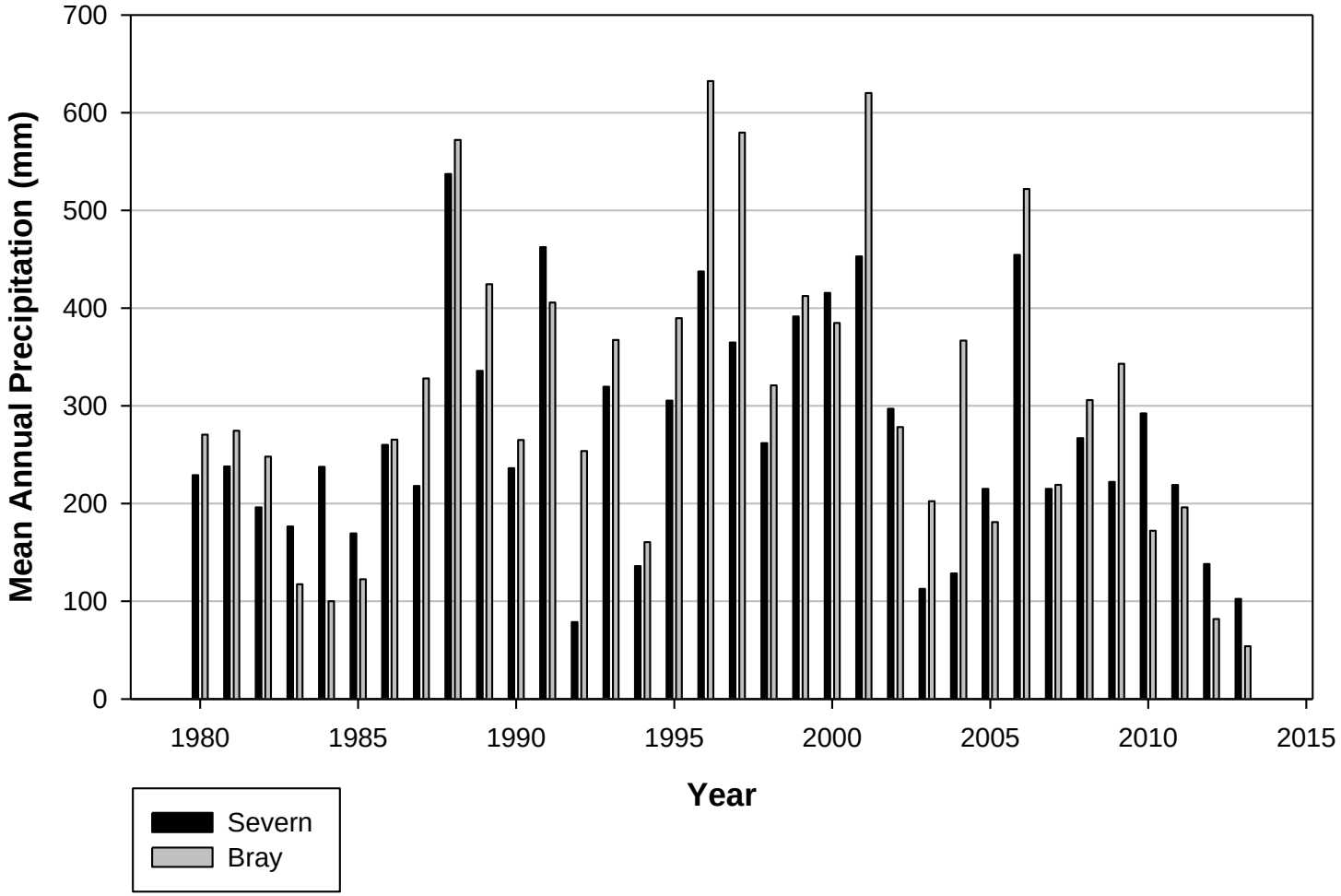

Figure 3.3: The mean annual precipitation for the 34-year-period from 1980 to 2013 at the weather station Severn [0428635 1] and Bray [0541297 5] (data source South African Weather Services, 2013).

3.3 Geology, pedology, topography and land types

A large part of the Molopo is situated in a rock formation unit of the larger Kaapvaal Craton that is a mafic intrusion related to the Bushveld Complex known as the Molopo Farms Complex (Walker *et al*., 2010; Prendergast, 2012). The Molopo Farms Complex is a large layered intrusion, approximately 13 000 km^2 in area (Prinsloo, 1994; Gabrielli, 2003). Presently, the complex is totally hidden beneath Karoo sediments and Kalahari sand in south-western Botswana and the northern parts of the North-West Province (Reichhardt, 1994; Anhaeusser, 2006). After bore-drilling, it was found that the Molopo Farms Complex is approximately 3 000 m thick and intrudes into sedimentary and minor volcanic rocks of the Transvaal Supergroup (c. 2650-2200 Ma), while it is elongated in a northeast-southwest direction, completely covered by up to 220 m of Cenozoic Kalahari Group sediments (Scholes *et al.*, 2002; Anhaeusser, 2006; Walker *et al*., 2010).

Red and yellow sandy, well-drained soils with a high base status (Arenosols AR2) dominate the Molopo (AGIS, 2010). Sand dunes occur closer to the drainage valleys such as the Molopo River (Van Niekerk, 2011). The Gordonia Formation, informally termed Kalahari sand, is a deep, red aeolian sand of recent age with surface calcrete, silcrete and ferricrete (Low & Rebelo, 1996; Rutherford *et al*., 2006) that covers most of the underlying Kalahari Group sediments (Partridge *et al*., 2006). However, in certain areas, it rests directly on the pre-Kalahari bedrock.

The soils are more than 1.2 m deep and sandy (Hutton and Clovelly soil forms) (Rutherford *et al*., 2006). Sandy soils are well permeable to air, water and roots but have a low water holding capacity and a low soil nutrient storage capability (Sims, 1981; Gebremeskel & Pieterse, 2006). The sand with its very low moisture retention abilities is not seen as a very suitable growth medium for vegetation but do, however, have the ability to retain some moisture during rainfall episodes. The moisture in the top six meter layer of the sand and the ability of certain woody plant species' root systems to penetrate underlying water tables ensure vegetation establishment in this semi-arid environment of the south-eastern Kalahari (Thomas & Shaw, 1991).

The topography of the study area varies between 1 000 and 1 300 m above sea level and is characterised by undulating to flat sandy plains with no permanent rivers

(Donaldson & Kelk, 1970; Rutherford *et al*., 2006). The area experiences little to no water runoff on a large scale due to the relative level topography and also because the larger part of the Molopo area is covered by Kalahari sand that has a high infiltration rate (Donaldson & Kelk, 1970).

3.4 Vegetation

3.4.1 Savanna Biome

The Molopo, situated within the south-eastern parts of the Kalahari, forms part of the Savanna Biome of Southern Africa (Low & Rebelo, 1996; Rutherford *et al*., 2006). The Savanna Biome is the largest biome of the southern African subcontinent and comprises 32.8% of the surface area of South Africa, which relates to 399 600 km^2 (Figure 3.1) (Rutherford *et al*., 2006). The term savanna describes a physiognomic type of vegetation with an herbaceous, usually graminoid understory layer and an upper layer of scattered woody plants (Trollope *et al.*, 1990; Rutherford *et al*., 2006), which can vary from widely spaced to 75% canopy cover (Rutherford & Westfall, 1986) (compare chapter 2 for a general description of vegetation dynamics in southern African savanna ecosystems).

In South Africa, this biome is mostly found at altitudes below 1500 m but do extend to 2000 m above sea level (Low & Rebelo, 1996; Rutherford *et al*., 2006). Rainfall patterns, nutrient availability, fire and herbivory are all key determinants of the vegetation structure and composition of savannas (Scholes *et al*., 2002). Many genera and species of the savannas of southern Africa are shared with the savannas of Central- and East Africa and can also be found in the Nama-Karoo Biome of southern Africa (Scholes, 1997).

Plant species richness of the Savanna Biome in southern Africa at a whole-biome scale is relatively high compared to the other southern African biomes with only the Fynbos Biome's plant species richness being higher. However, per unit area, the plant diversity of savannas is lower than in the Fynbos, Forest, Grassland or Succulent Karoo Biome (Scholes, 1997). The Savanna Biome contains 3-14 species m^{-2} and 40-100 species 0.1 ha^{-1}, not significantly different to the rest of southern African biomes (Scholes, 1997).

3.4.2 Kalahari vegetation

In a study conducted by O'Brien (1993) to determine the diversity of woody plants over southern Africa, the author found the species richness being the lowest in the Kalahari Basin. Species richness increased towards the eastern escarpment. A strong positive relationship between the woody plant species richness and the mean annual precipitation (MAP) was found for the Kalahari region (O'Brien, 1993). Scholes *et al*. (2002) found a strong positive relationship between the woody plant basal area and MAP along an aridity gradient in the Kalahari. Woody biomass, cover and height also tend to increase with an increasing MAP. As the water availability decreases, woody plant densities become sparser and also lower in height. \

There are many factors contributing to the distinction between savannas, but the main functional distinction between the Savannas of southern Africa is broad- and fine-leaved savannas. Fine-leaved savannas occur in nutrient-rich, arid environments and the broad-leaved savannas in nutrient-poor, moister environments (Scholes, 1997), an exception being broad-leaved *Colophospermum mopane* savannas that occur in fertile arid savannas (Scholes, 1997). Fine-leaved savannas contain relative palatable grass species, attracting high ungulate densities preventing fires as the grass biomass is constantly being removed. Broad-leaved savannas can be distinguished from fine-leaved savannas by having less palatable grass species, fewer ungulates and more fires (Scholes *et al*., 2002).

3.4.3 Molopo Bushveld

Acocks (1988) described only one veld type for the Molopo study area, namely the Kalahari Thornveld which covers the entire study area. It is similar to Low and Rebelo's (1996) Kalahari Plains Thorn Bushveld and Rutherford *et al*.'s (2006) Molopo Bushveld vegetation type. The tree layer of this vegetation type is dominated by *Acacia erioloba* and *Boscia albitrunca*, with less dominant species being *A. luederitzii* and *Terminalia sericea* (Scholes *et al*., 2002*;* Rutherford *et al.*, 2006). The shrub layer is moderately developed with species such as *A. mellifera*, *A. hebeclada*, *Lycium hirsutum*, *Grewia flava* and *A. haematoxylon*. The grass layer is dominated by perennial grasses such as *Aristida meridionalis*, *A. stipitata, Eragrostis lehmanniana*, *E. pallens*, *Stipagrostis uniplumis, Schmidtia pappophoroides* and *Urochloa mosambicensis* (Richter *et al*., 2001). Also present within this vegetation

type are annual grasses such as *S. kalahariensis* (Low & Rebelo, 1996; Rutherford *et al*., 2006). Species endemic to the Kalahari region that occur in the Molopo Bushveld vegetation type are the small tree species *A. luederitzii var. luederitzii* and *A. haematoxylon*, which is a tall growing shrub. Endemic grass species include *Anthephora argentea, Megaloprotachne albescens* and *Panicum kalaharense* (Rutherford *et al*., 2006).

The Molopo Bushveld vegetation type covers parts of the North-West and Northern Cape Provinces of South Africa (Figure 3.1). The area ranges from open woodland to a closed shrubland with a well-developed grass herbaceous layer in the north-eastern parts but is usually fairly open (Rutherford *et al*., 2006). The approximate long-term grazing capacity for this vegetation type is 10 ha LSU^{-1} (Large Stock Unit) (Mostert *et al.*, 1971 *in* Richter *et al.*, 2001).

This vegetation type is categorised as least threatened for conservation. Only 1% of the targeted 16% of the Molopo Bushveld is conserved in the Molopo Nature Reserve, and more than 1% has already been transformed. Over-utilisation by livestock has led to the thickening of the woody plant *A. mellifera* subsp. *detinens*, and a lot of the *A. erioloba* has been destroyed for fire-wood purposes. The erosion extent is very low (Rutherford *et al*., 2006).

4

Integrated Assessment of Restoration and Management Actions

4.1 Introduction

A need exists for the evaluation of management and restoration actions to combat rangeland degradation with a view to providing essential inputs for decision making (UNCCD, 2009). Most assessments of dryland management systems, such as rotational grazing management, focus on the biophysical properties of the system (e.g. soil erosion and grass cover). It is, however, important to also include socio-ecological assessments, taking the financial and cultural challenges of the land user into account (Reed *et al*., 2007). The incorporation of local indigenous knowledge in the assessment process of environmental problems and potential mitigation actions have also been increasingly called for by international institutions such as the United Nations Convention to Combat Desertification (UNCCD, 2009).

To this end, PRACTICE (Prevention and Restoration Actions to Combat Desertification: An Integrated Assessment) brings together scientists and stakeholders (SHs) from regions all over the world affected by land degradation and desertification to learn from one another's experiences in combating desertification by way of afforestation, improving pastures, controlling grazing, managing watershed and taking long-term sustainable agricultural actions (Rojo *et al*., 2012). The main objectives of PRACTICE is (a) to fill the gap in terms of the systematic evaluation of actions for rangeland restoration and sustainable land management in a participatory and integrated manner, (b) to link science to society with a view to the sharing and transferring of evaluation methods and actions to combat rangeland degradation and (c) to combine local (indigenous) knowledge with scientific expertise. In pursuit of these objectives, PRACTICE aims to develop and implement an integrated evaluation protocol in order to assess the effectiveness of different restoration and

management actions implemented to combat desertification that is applicable worldwide (Rojo *et al.*, 2012).

In this study, the Integrated Assessment Protocol (IAPro), which forms part of the PRACTICE programme, was applied (Bautista & Orr, 2011) in order to achieve the following aims: (1) to identify restoration- and rangeland management actions implemented by local land users and communities affected by rangeland degradation, (2) to evaluate the performance and acceptance of restoration actions in an integrative way, (3) to share knowledge and results with a multi-stakeholder platform (MSP) and (4) to foster the implementation of best actions in a locally contextualised manner. The IAPro is based on key indicators that represent overall ecosystem and human-environmental system functioning, site-specific indicators that are identified by local stakeholders (SHs), the perspectives of SHs and biophysical measurements for indicator quantification (Rojo *et al*., 2012).

4.2 Material and methods

4.2.1 Integrated assessment protocol

The IAPro follows a range of sequential steps (Figure. 4.1) – four of the steps being fully participatory – that offer a path for the exchange of knowledge among a variety of SHs (e.g. scientist, rangeland managers and local land users) (Bautista & Orr, 2011). Steps 1 to 3 and 6 are all full participatory activities' involving the SHs. Step 4 involves only the scientific data captured by the local assessment team, and step 5 integrates the scientific data with the data captured during the participatory activities involving the SHs. The process of combining local perspectives of SHs with biophysical and socio-economic assessment data is referred to as the participatory process (Bautista & Orr, 2011).

The final step of the IAPro (step 7) focuses on participatory activities and, in this instance, will involve distributing the knowledge learned amongst the respective local SHs from the Molopo study area (as identified in this project) during the evaluation process. This knowledge will be shared with the larger global community that is affected by desertification and rangeland degradation through internet-supported dissemination of audio-visual media (Bautista & Orr, 2011; Rojo *et al*., 2012). Thus,

only the first 6 steps of the IAPro have been completed in the course of this study and will now be reported on.

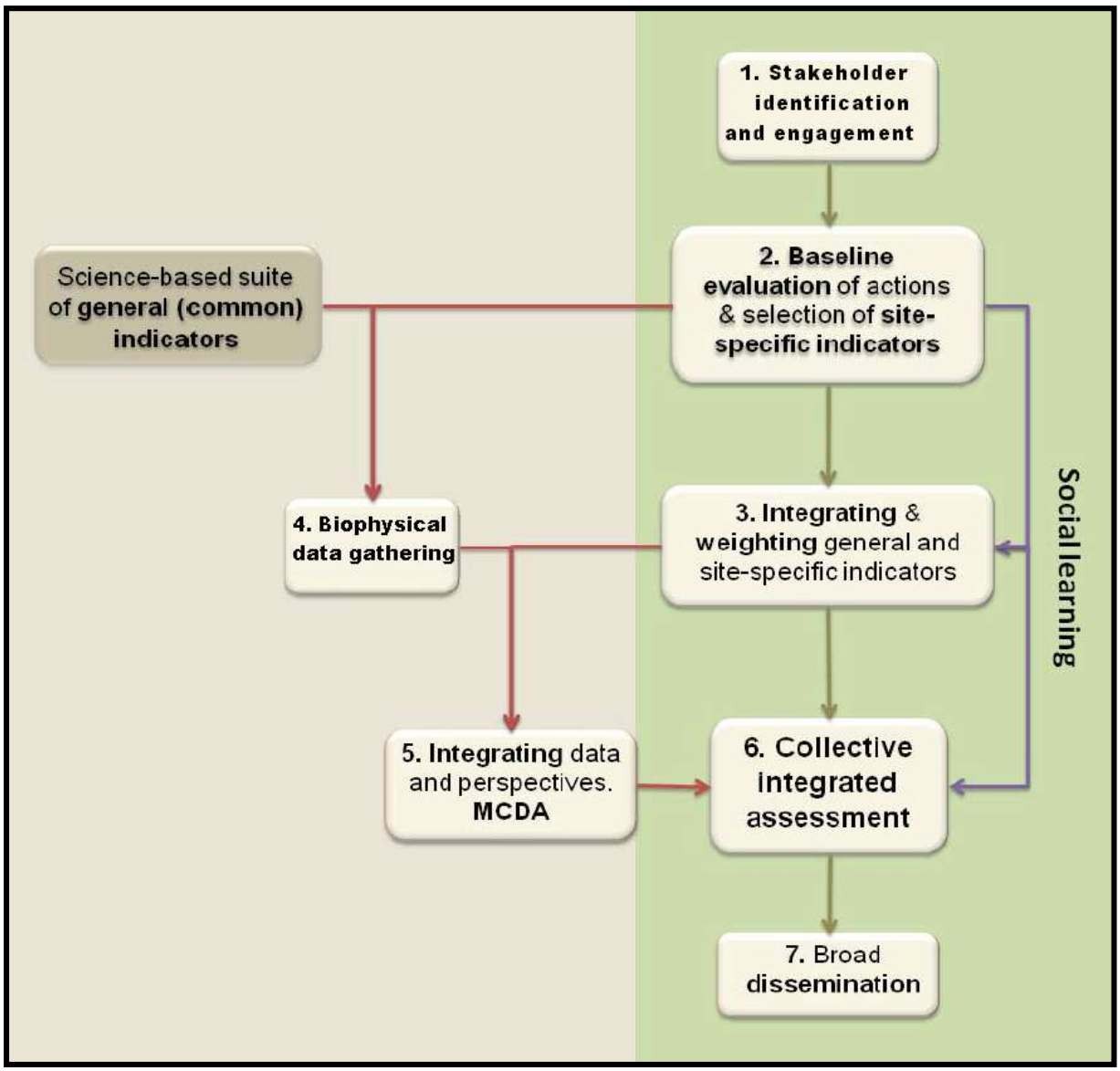

Figure 4.1: The IAPro structure indicated by a flowchart of the protocol steps (source: Bautista & Orr, 2011).

Step 1 & 2: Identification of local SHs to set up a multi-stakeholder platform followed by a baseline evaluation of restoration and management actions (step 2)

Due to time constraints and the distance involved to travel to the study area, step 1 and 2 were conducted simultaneously. The interviews for step 1 and 2 were conducted from August 2010 to August 2011.

In IAPro step 1, a representative group of various SHs was identified, the latter being able to contribute to the process of evaluating restoration and management actions applied in the Molopo area. Individual, semi-structured interviews were conducted with locally known land users, extension officers and rangeland scientists as well as through a local consultation process and chain referrals (i.e. referral of other potential SHs who could participate in the study). With the aid of a stratified, purposive and opportunistic sampling method (Bernard, 2006), a MSP could be established for involvement throughout the project. For this purpose, 60 interviews were conducted, and ultimately 46 SHs were identified to take part in the study based on the following criteria: (a) agricultural expertise, (b) experience and action of rangeland management and restoration, and (c) willingness to participate in the project.

The interviews were conducted either in the home of the land user, in an office or outside under a tree where the SH felt more comfortable. Before each interview, a description of the research project was shared with the SH and also the nature of participation. The interview was guided by questions compiled by Dr Taryn Kong based on the guide on questions provided by Bautista and Orr (2011) (Data sheet A1).

The interviews were viewed more as a discussion taking place between the interviewer and SH. Nevertheless, a number of key topics were considered to structure the interviews: (a) social position (i.e. occupation and responsibilities) of each SH, (b) the SH's perspectives on general socio-environmental conditions for the study site, (c) the SH's knowledge of restoration and management actions, (d) the SH's level of interest and potential elements that could prevent the SH from participating in the process, (e) referral of other potential SHs that could participate in the process and (f) basic information on the respondent such as age, gender and

preferred language. Additional topics for possible discussion were also raised by the interviewer with a view to keeping the conversation going and to ensure that the SH felt comfortable.

Each 60 – 90 minute interview was audio-recorded, and some of the interviews were conducted in the native language (Setswana and/or Afrikaans). The latter had to be translated into English by Mr Isaac Makwana and Mr Joseph Nkoko from the Department of Agriculture and Rural Development (DARD), North-West Province (Setswana) and Mr Christiaan Harmse and Ms Marguerite Westcott from the North-West University, Potchefstroom Campus (Afrikaans).

The local assessment team for the interviews included Dr. Taryn Kong from the University of Arizona, Mr Christiaan Harmse and Ms Marguerite Westcott from the North-West University, Potchefstroom campus, and Prof Franci Jordaan and Mr Ernst Mokua from the DARD, North-West Province. The personal contribution of Mr Christiaan Harmse in the first two steps included (a) acting as note taker and translator and (b) accommodating land users on field trips to the rangelands to identify and make notes on the coordinates of the areas where restoration and management actions have been implemented.

During step 2 of the IAPro, the 46 SHs were encouraged to elaborate on their livelihoods and to identify baseline SH perspectives regarding (1) various rangeland management systems and restoration actions applied to combat rangeland degradation (e.g. rotational grazing systems or chemical bush control) and (2) site-specific indicators (criteria) used to assess the effectiveness of the respective restoration and management actions. Taking into account aspects of redundancy, popularity, availability of data and collectability of the indicators proposed in step 2, combined with expert common indicators based on ecosystem services (such as grass diversity, grass phytomass production, grass cover and soil organic carbon), a condensed list of indicators was compiled during this step. This data was used for the evaluation of the restoration and management actions mentioned in step 2 of the IAPro.

Step 3: Indicator weighting exercise

Step 3 aimed at establishing both individual and integrated SH perspectives on the relative importance of the final set of indicators selected in previous steps to assess the management and restoration actions to combat rangeland degradation. This step was designed to collect information on SH preferences on both the site-specific indicators identified by the local SHs and the science-based common indicators that represent the overall function of dryland ecosystems. The step involved two phases: (a) individual baseline weighting of the final set of indicators and (b) integrated collective weighting. These weighting processes (each iteration) were undertaken by the SHs attending two workshops that were held after steps 1 and 2 (see steps 3a and 3b below). This was the first opportunity within the IAPro for social learning and the integration of scientific and local knowledge between land users who participated in the interviews.

Step 3a: Weighting of final set of indicators

Three months after the first two steps have been conducted, the same local assessment team who conducted step 1 and 2 revisited the SHs to conduct the first part of the IAPro step 3 (i.e. step 3a) during the first workshop. Step 3b and the second workshop followed after the quantitative assessment in step 4.

In step 3a, the relative importance value of each indicator for assessing the mitigation actions based on individual SH perspectives was established (first iteration). This was done by 38 of the participants identified in step 1. After the computation of the weighting results for the indicators, the indicators were ranked from the most to least important (Data sheet A2).

To assign a weight value to each indicator, the Simons' pack-of-cards approach (Figueira & Roy, 2002) (Figure 4.2 and Figure A1, in Appendix) was applied and the results computerised in accordance with Figueira and Roy (2002). Each indicator was assigned to a colour card by way of an illustration and explanation on the back of the card. SHs organised the cards (indicators) in different importance levels, and blank cards (black cards in Figure 4.2) could be used to reinforce and distinguish between the different levels of importance of each indicator.

For example, if one black card was placed between two indicators, the second indicator (H – Figure 4.2) was twice as important as the previous one (D - Figure 4.2) (Bautista & Orr, 2011). When indicators were similar in importance, the cards (representing the different indicators) were placed beneath one another (Figure 4.2). The absolute value of each card and blank card was unknown at this point in time, since there is no specific numeric scale for the overall ordering.

Additionally, participants were asked to provide an estimation of the ratio of difference between the most and least important indicator, called the Z Ratio (Bautista & Orr, 2011). This ratio was only determined by the participant and could have been any number, e.g. 2, or 10, or 50. If the participant decided on 2, it would mean the participant is of the opinion that the most important indicator is twice as important as the least important indicator. This ratio number-scaled the distribution of the resulting weights in the computation software SRF (see Figueira & Roy, 2002). The computation exercise was conducted for each SH separately, and the individual weights were then integrated using a simple average per indicator to provide a baseline set of collective weights from all the SHs.

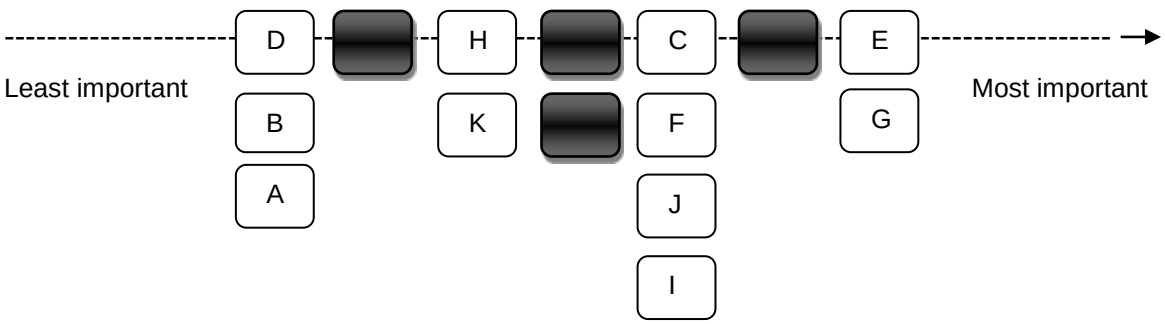

Figure 4.2: Outline of a possible arrangement of indicators together with blank cards (black cards) during a weighting exercise (source: Bautista & Orr, 2011).

Step 3b: Discussing individual and collective results and re-assessing the indicators

In step 3b, the aim was to establish integrated SH perspectives (second iteration) during a second workshop. The first weighting of the individual SH perspectives in the first iteration (step 3a), as well as the biophysical data sampled during the quantitative surveys (step 4), was presented to the multi-stakeholder platform during

this workshop held in June 2013. Twenty of the participants identified in step 1 attended the workshop.

The multi-stakeholder platform was divided into small groups who were then encouraged to discuss each other's and also the integrated group's results, thus promoting the exchange of knowledge and contributing to social learning. This was followed by an open group discussion taking the results from the first iteration and the biophysical results from step 4 into consideration. It must be noted that results from the socio-economic indicators were not presented to the SHs as these indicators were either not directly measureable (e.g. type of risks that are applicable, such as fire and drought) or could not be assessed due to land-user confidentiality (e.g. income and profit).

Following on the second workshop, the indicators were re-ranked (second iteration) to determine whether the perspectives of the SHs had changed after having been presented with the group results and the results obtained from the biophysical assessments and the first workshop. A final set of weights was then calculated for each indicator. The Mann-Whitney-Wilcoxon test was applied, using PAST v.2.15 (Hammer *et al*., 2001) to determine significant differences between the relative importance values for the indicators averaged over individual SH perceptions before (first iteration) and after (second iteration) group discussions. Differences between the two iterations in these steps were used as metrics of the social learning between the land users after the workshops.

Step 4: Biophysical data gathering

Site selection for step 4 followed a preferential sampling design guided by the local land users and was based on information gathered from the semi-structured interviews in step 1 and 2 of the IAPro regarding the SHs' perceptions of degraded land and what was successfully restored or better managed land. Criteria used for site selection by the land users included, amongst others, grass species composition, grass phytomass production, abundance and composition of woody species and accessibility for livestock or game. Six sites were sampled on communal farms around Eska Neuham and Kgokgojane, 10 sites on lease cattle farms around Ganyesa and Vostershoop, 23 sites on commercial cattle farms near Severn,

Vostershoop and Bray and six sites on a commercial game farm near Bray (Figure 4.3).

Quantitative assessments were carried out on each of the sites to capture data for each of the indicators identified in the final set of step 3a, which included both general and site-specific indicators. The quantitative assessments on the woody and grass layer were carried out from January to March 2012 and from February to March 2013. In November 2012, only woody surveys were conducted.

Data gathering was handled by a team of rangeland ecologists from the NWU with the participation of SHs on their farms. Here it ought to be noted that no initial quantitative surveys of the vegetation were conducted before restoration and management actions have been implemented at each of the selected sites.

The data of the quantitative surveys carried out for this project in the areas where the restoration and management actions as a mitigation action to combat land degradation/desertification have been applied were compared to controlled areas (n = 14) on the same rangelands where no mitigation actions have been applied. The controlled sites were usually characterised by densely bush-thickened areas (Figure A7, in Appendix) with a low rangeland productivity in terms of grazing. Long-term rotational grazing management sites (n = 8) served as the benchmark for the restoration and management actions implemented as mitigation actions due to the higher rangeland productivity measured in these areas (Figure A11, in Appendix).

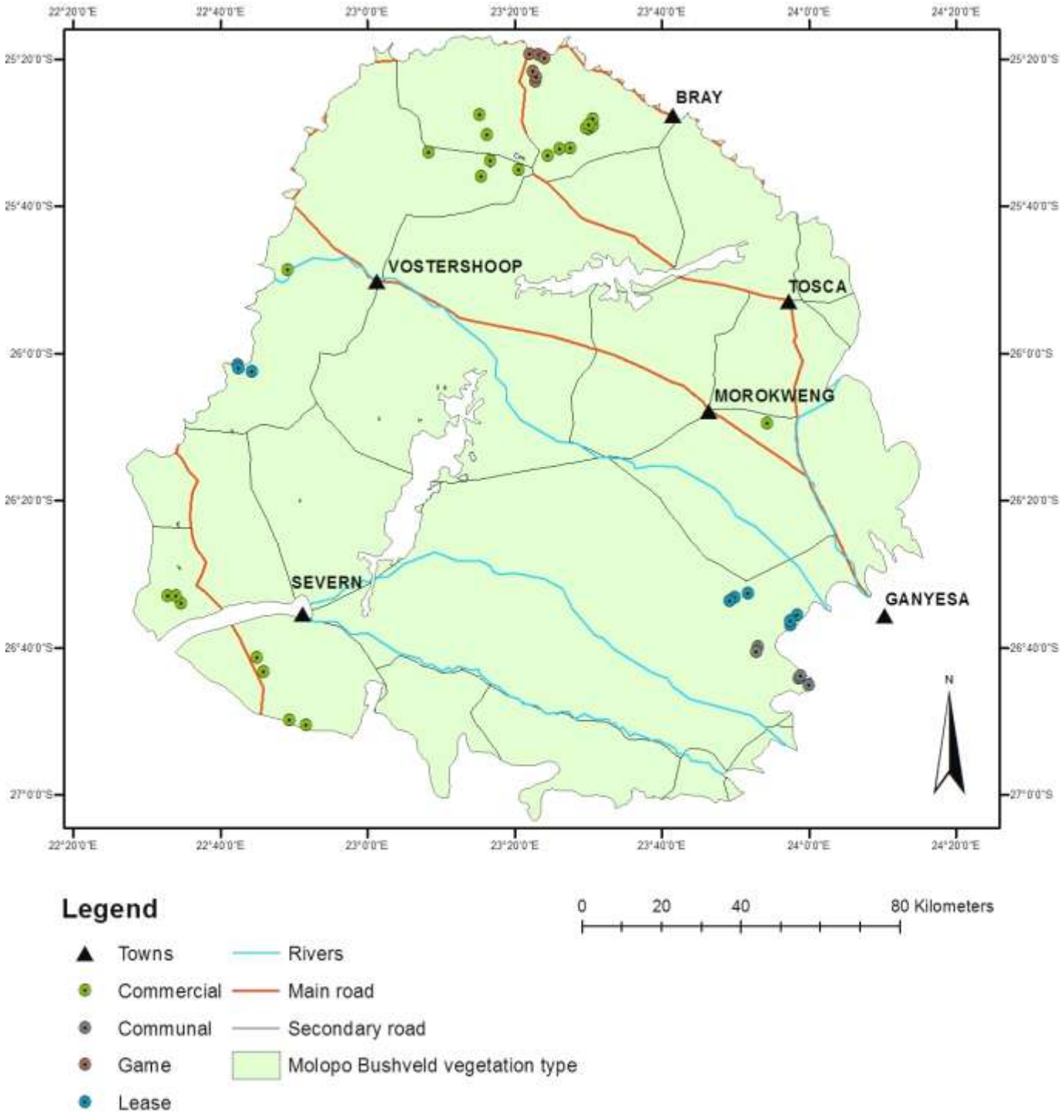

Figure 4.3: The spatial distribution of the 50 vegetation sampling sites over four different land tenure systems (commercial, communal, game and lease) indicated within the vegetation type (i.e. Molopo Bushveld *sensu* Rutherford *et al.* (2006)).

Step 4a: Biotic and abiotic environmental indicators

The final set of indicators identified in step 3a of the IAPro by the SHs related to rangeland productivity and biodiversity was quantitatively assessed using the fixed-point monitoring of vegetation (FIXMOVE) methodology (Morgenthal & Kellner, 2008). The FIXMOVE methodology was adapted from a previous monitoring method developed by Trollope *et al*. (2004) and entailed the following: The nearest rooted grass tuft according to the point method (Kent & Coker, 1996) and distance-to-tuft method (see Trollope *et al*., 2004) were used to monitor the grass layer (see section 4.2.2.1), and the woody layer was assessed with the aid of the adapted point-centred quarter method (APCQM) developed by Trollope (2011) (see section 4.2.2.2).

Other assessment parameters included the grass species composition, frequency, abundance and species richness of the grass layer, as well as the woody species composition and density. In addition, soil surveys were also performed at each assessed site. Afterwards, soil samples were analysed at the Department of Soil Sciences of the NWU, Potchefstroom[*], to determine the percentage organic carbon. Grass forage production and grazing capacity were also calculated (see section 4.2.2.1) to quantify certain indicators identified by SHs such as grass forage production and grazing capacity.

Additional information recorded at each site included site identification name, date of survey and GPS reading (Table A1), as well as a photograph documenting the vegetation and habitat of each survey site.

Step 4b: Social and economic indicators

As mentioned above, social and economic indicators were either not directly measureable (e.g. risks) or could not be assessed due to land-user confidentiality (e.g. income and profit). Therefore, SHs were asked during the second workshop to rank the mitigation actions and the impact thereof regarding the social and economic indicators. Actions were ranked from 1 to 5 with 1 having the most positive influence to 5 having no or the least positive influence on each of the indicators (Datasheet A4).

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Step 5: Integrating data and perspectives

A multi-criteria decision analysis (MCDA) is an approach that was developed to assist decision makers in solving problems where contradictory criteria and points of views often apply at the same time (Bautista & Orr, 2011). The MCDA using ELECTRE IS (Aït Younes *et al*., 2000) is mainly applied as an outranking procedure based on pairwise comparisons of all the different management and restoration actions. The result is then used to choose the best option among the set of alternatives, rank the set of alternatives from best to worst and/or sorting alternatives according to their performance (indicator quantification). The mean collective weight values of the final set of indicators elicited in step 3b were used in ELECTRE IS.

The MCDA makes use of thresholds and cutting levels from the ELECTRE IS software. In this study it was found that a lower value, for instance $\lambda = 0.80$, didn't clearly differentiate between the various restoration and management actions. Since the aim was to identify the most effective action to combat rangeland degradation, we made used of a higher cut level value (i.e. $\lambda = 0.90$).

Through the IAPro process, a MCDA was applied to integrate the SHs' perspectives on the assessment method and the biophysical and socio-economic measurements for each indicator measured in step 4. The final result of the procedure is the partial or ordinal ranking of options which helps to visualise and identify indicators that are either indifferent (performed equally well in both actions being compared) or supportive for a certain action over another action (e.g. Action 1 outranks Action 2 if for a large number of the criteria/indicators Action 1 performs at least as good as Action 2, see Figure A2 in Appendix) (Bautista & Orr, 2011).

Step 6: Collective integrative assessment

This step was conducted at the same time as step 3b during the second workshop held in June 2013 with 20 of the SHs from the multi-stakeholder platform in an open, interactive setting to encourage knowledge exchange and social learning through group discussions. The main aim of the last and final step (step 6) and overall goal of this study objective was to obtain a final evaluation of the identified actions (Bautista & Orr, 2011), which included the sharing of the results from the MCDA obtained in step 5 with the multi-stakeholder platform in a non-scientific language together with easily accessible figures and visualisations.

By reflecting on the results presented at both the workshops, SHs are stimulated to make more informed decisions regarding the implementation of restoration and management actions to mitigate land degradation/desertification and to encourage the re-evaluation (final evaluation) of the actions. As was the case in IAPro step 2, the SHs were asked to evaluate the actions on a Likert scale (Likert, 1932) from 1 (a very bad choice) to 5 (an excellent choice). All the identified actions were listed on a wall, and SHs were handed cards displaying the numbers 1 to 5. The SHs then placed the cards with numbers next to the name of the action onto the wall to indicate their choice of mitigation they would apply on their land ranked from bad (1) to excellent (5) (see Figure A3 in Appendix).

4.2.2 Methods and calculations for quantitative vegetation assessments

As mentioned above (see step 4a), the quantitative vegetation assessments were carried out following the fixed-point monitoring of vegetation (FIXMOVE) methodology (Morgenthal & Kellner, 2008) with certain adjustments. The vegetation assessments were carried out in the areas where the restoration and management actions have been applied and were then compared to a control area where no mitigation actions have been applied.

4.2.2.1 Grass layer

To assess the grass layer, two contiguous parallel 100 m-long transects were laid out 40 m apart in homogenous vegetation stands per sampling site. The starting point of the first transect was determined at random. The grass species composition surveys were all conducted during the rainy (growing) season of 2012 and 2013 (January to March) (See Figure 3.3). To determine the grass species composition, a rod was lowered directly next to the transect line at 1 m intervals (Figure 4.4) (Morgenthal & Kellner, 2008). The nearest grass plant to the point of the rod was identified up to genus level, and the distance to the plant was measured. In case no grass plant was present within a radius of 30 cm, bare ground was recorded. If the plant could not be identified during the assessment, it was taken to the A.P. Goossens herbarium at the Potchefstroom campus of the NWU[†].

[†] Goossens, A.P., Herbarium (North-West University, Potchefstroom campus)

The disc pasture meter (DPM; Bransby & Tainton, 1977) was used to assess the grass phytomass production at each survey site. Trollope and Potgieter (1986) already calibrated the DPM previously for the use thereof in the southern African savannas and they found a very low variation across the vegetation types in the Kruger National Park. A DPM measurement was collected at 1 m intervals on the left-hand side of the transect 1 m away (Figure 4.4) (Morgenthal & Kellner, 2008). The DPM is based on the assumption that a similar amount of standing grass biomass settles at a similar height when the round aluminium disk is dropped along a pole from a fixed height (see Figure A4 in Appendix for an example of a DPM) (Morgenthal & Kellner, 2008). A quadrate (1 m × 1 m) was placed directly on the right-hand side of the line transect every 10 m (Figure 4.4) to determine the abundance of each grass species by counting the number of individual grass tufts and seedlings per quadrate (Kent, 2012) in order to quantify the indicator “grass abundance”.

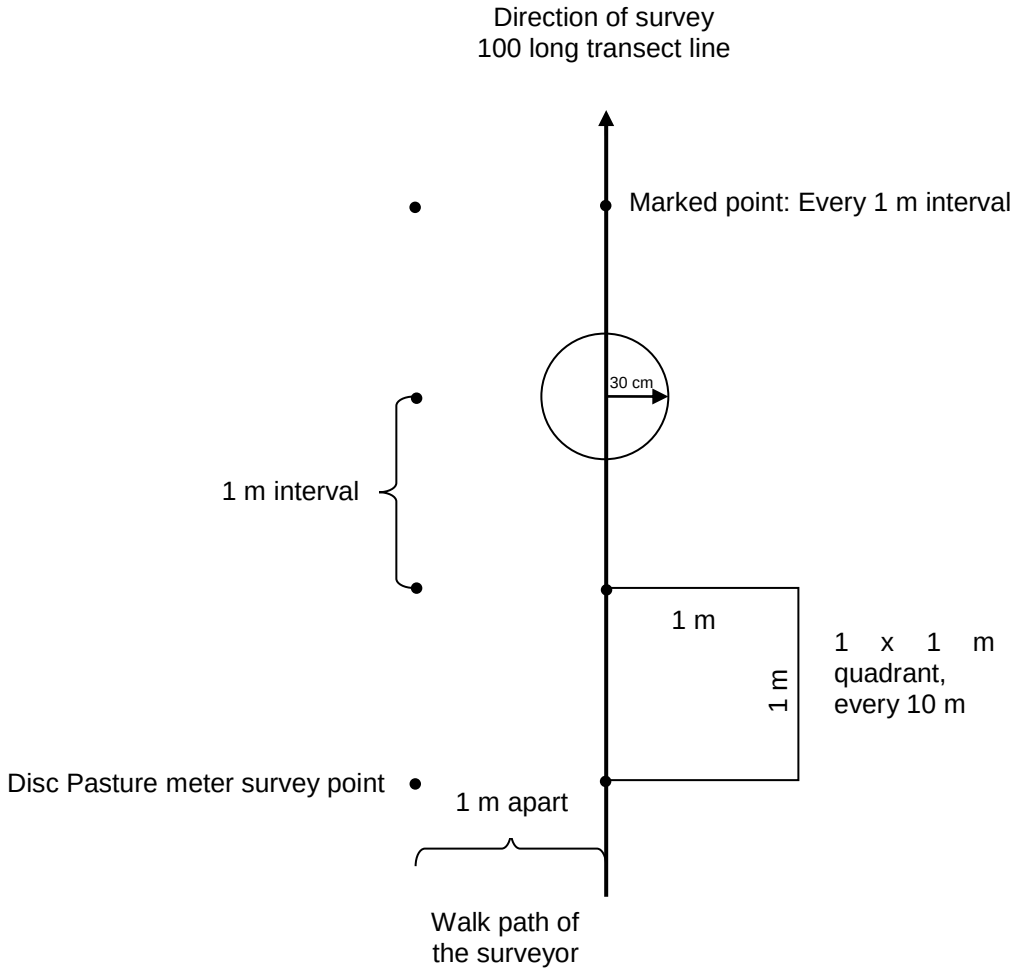

Figure 4.4: Illustration of the grass layer survey carried out along one of the two 100 meters transects.

Calculations

The following data was calculated based on the quantitative results obtained from the FIXMOVE methodology for the grass layer: (a) diversity of grass species (Shannon-Weiner diversity index) to quantify the indicator "grass diversity", (b) dry matter grass production (DM) from the DPM (kg ha^{-1}) for the "forage production" indicator, (c) grazing capacity (ha large stock unit $(LSU)^{-1}$) and (d) the "grazing capacity" indicator that was quantified from the palatability and grazing value (high, moderate and low) of the different grass species according to Van Oudtshoorn (2012).

The grazing capacity was expressed in ha LSU^{-1} according to Moussa *et al*. (2009) using the formula 365 days × 10 kg / *DM* × % grass palatability, where 365 is the

total days of the year, 10 kg the total dry matter (*DM*) yield utilised by a large stock unit (i.e. a cattle of 450 kg) day^{-1}, *DM* the total leaf dry matter yield ha^{-1} as measured by the DPM and percentage palatability (Van Oudtshoorn, 2012) of the available grass material, i.e. high palatability (35%), moderate palatability (20%) and low palatability (5%) (Jordaan, personal communication‡). A conservative grazing capacity value of only 60% available DM material, including all the grass species, was considered during the calculation. The composition was determined by the nearest rooted grass tuft point method and DM value with the DPM. A palatability class (Van Oudtshoorn, 2012) was then assigned to each grass species in order to calculate the % grass palatability before the total DM was determined as used in the above formula.

Grass diversity was expressed by species richness (i.e. number of different grass species) and the Shannon-Weiner diversity index. The Shannon-Weiner diversity index (H') was calculated as $H' = - \sum p_i \log p_i$ (MacArthur & MacArthur, 1961), where *s* is the number of species, p_i the proportion of individuals or the abundance of the *i*th species.

4.2.2.2 Woody layer survey

The woody layer survey was conducted by making use of the adapted point-centred quarter (PCQ) method (Trollope, 2011) on the same two transects used for the grass layer survey. The two transects were, however, extended from 100 m to 160 m in length. Each transect had a total of four recording points per sample site and a spacing of 40 m between recording points. Each recording point was subdivided into quarters with side lengths of 20 m (Figure 4.5) (Trollope, 2011).

In each quarter, the following data were recorded for the nearest rooted woody plant from the recording point in three different height classes (i.e. < 2 m, > 2 m and tallest woody) (Figure 4.5), i.e. the type of species, the distance from recording point and certain biometric characteristics (see below) to determine the tree equivalents (TE) ha^{-1}. The distance to the centre of the woody plant was recorded in the case of multi-stemmed plants. According to Trollope (2011), only woody plants growing within a

‡ Jordaan, F. Department of Agriculture and Rural Development (DARD) - North West Province, South Africa. Contact details: Telephone: +27 (0)18 299 6702.

radius of 20 m from the recording point should be sampled. However, it was found more practical to consider the area of a whole quarter (20 × 20 m) with an observer at the outer fringes deciding if a plant was inside or outside the quarter.

Biometric measurements for each woody plant were taken according to Smit (1989[a]): (1) height of the woody plant, (2) height of maximum canopy diameter, (3) height of lowest leaves (browseable material), (4) two maximum canopy diameters that are rectangular to each other and (5) the basal canopy diameter. Plastic poles that were 2 m long and marked in 10 cm intervals were used for biometric measurements at each woody plant (Figure A5, in Appendix). For woody plants taller than 2 m, the pole was placed next to the plant and the dimensions were then estimated from a distance.

In this open savanna vegetation, there were often no woody plants in the quarter of the PCQ. In these instances, no plants were recorded (Trollope, 2011). However, this might have led to an overestimation of woody plants per hectare as the average recording distance is lowered. In order to prevent this, woody plants also outside the quarter to the left or right up to a distance of 30 m were considered. Respective plants were marked to prevent re-sampling when conducting the survey on the second transect, which was parallel to the first.

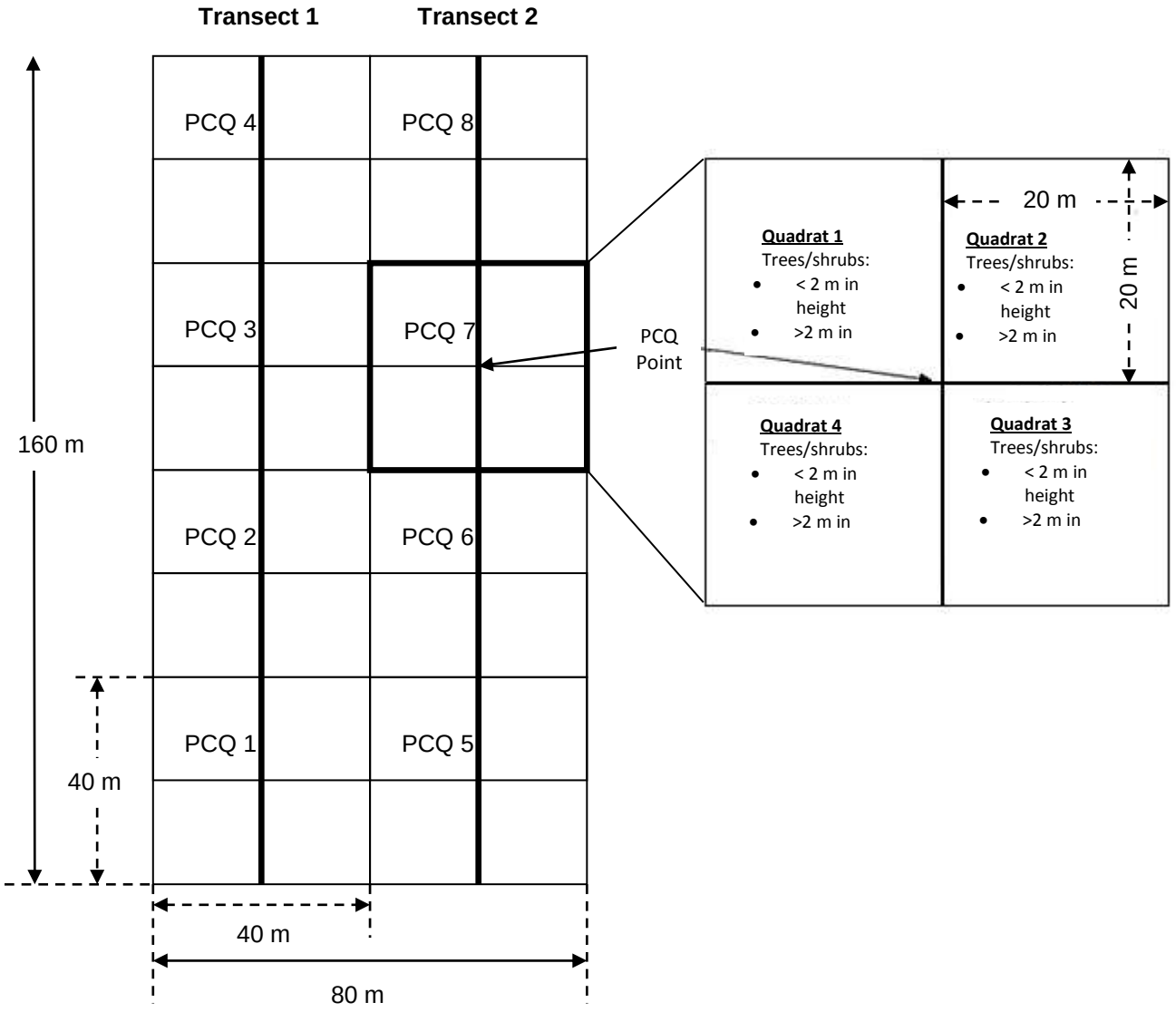

Figure 4.5: Graphical representation of the adapted PCQ methodology (Trollope, 2011) used for sampling the woody layer at the sampling sites.

Calculations

Woody density was calculated based on the formula 10 000 m^2 / D^2, with D^2 being the distance of the base of the woody plant from the recording point expressed in square meters (Cottam & Curtis, 1956). For each height class, the density was calculated separately, with the total woody plant density being the sum of the mean densities of the three height classes (Trollope, 2011).

The phytomass of woody plants relates to the degree of shrub thickening and was expressed as tree equivalents (TE) ha^{-1}, defined as a plant 1.5 m high (Teague *et*

al., 1981). The tree equivalents were calculated for all three height classes by $n \times h \times 1.5^{-1}$, with n being the total number of woody plants ha^{-1} for the height class and h the mean height for the woody plants within the same height class. Total phytomass is the sum of TE ha^{-1} calculated for the three height classes (Trollope, 2011).

Additionally to the quantitative vegetation surveys, soil samples were collected from the 30 cm top soil layer at all the survey sites by making use of a manual soil auger (Figure A6, in Appendix). Nine random samples were collected along the two 160 m transects. Out of the nine random soil samples, soil material was taken and pooled together, thoroughly mixed and divided again to form three composite samples of approximately 1 kg. Before the samples were sent to be analysed at the Department of Soil Sciences of the NWU, Potchefstroom[§], the three composite samples per site were mixed together in order to gain one representative sample. The soil pH was tested during the saturation extract, in a potassium chloride (KCl) solution and in a water (H_2O) solution

4.2.2.3 Statistical analysis

Boxplots were primarily examined prior to an analysis of variances to determine whether there were any outliers and if data log-transformation was necessary (Quinn & Keough, 2002). Differences among the restoration and management actions for each biophysical and socio-economic indicator were analysed using one-way ANOVA (Hammer *et al*., 2001). The post-hoc Tuckey's honestly significant difference (HSD) test (Hammer *et al*., 2001) was then performed to determine and compare the statistical differences between the various actions. All differences were considered significant at $P < 0.05$, unless otherwise stated. To achieve homogeneity of variances, grass species diversity was log-transformed. All analyses were carried out using PAST v.2.15 (Hammer *et al*., 2001).

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4.3 Results

4.3.1 IAPro step 1: Stakeholder identification

Out of the 60 interviews conducted during Step 1 of the IAPro, a group of 46 local stakeholders (SHs) was identified to make up the multi-stakeholder platform (MSP). The professional backgrounds of the members of this MSP differs, with land users of varying tenure systems (i.e. communal, lease and commercial) forming the majority (69.6%) in the MSP, followed by governmental experts (21.7%), service providers, academics and conservation personnel (Table 4.1; Figure 4.6). The SHs were predominantly male; nevertheless, two female land users originating from small-scale LRAD (Land Redistribution for Agriculture Development) farms did partake in the project. The group of land users comprising the MSP consisted of 32 livestock farmers, one commercial game farmer and one game reserve manager. The MSP further included five extension officers from the North-West Province's Department of Agricultural and Rural Development (NW-DARD), two consultants and six researchers (Table 4.1; Figure 4.6). The land-user categories provided in Table 4.1 are further explained in Table A2.

Table 4.1: Composition of the multi-stakeholder platform (MSP) identified through a local consultation process and chain referrals in the Molopo area of the North-West Province: The total number of participants in each category is indicated by way of figures.

	Type of expertise				
	Land user	**Governmental expert**	**Service provider**	**Academic**	**Conservationist**
Stakeholder category	- commercial-private (9) - semi-commercial-lease (4) - small-scale communal-tribal (11) - small-scale LRAD[§§] (8)	- extension officer (5) - researcher (5)	- consultant in rangeland and livestock management (2)	- researcher in rangeland ecology (1)	- manager (1)

[§§] LRAD = Land Redistribution for Agricultural Development

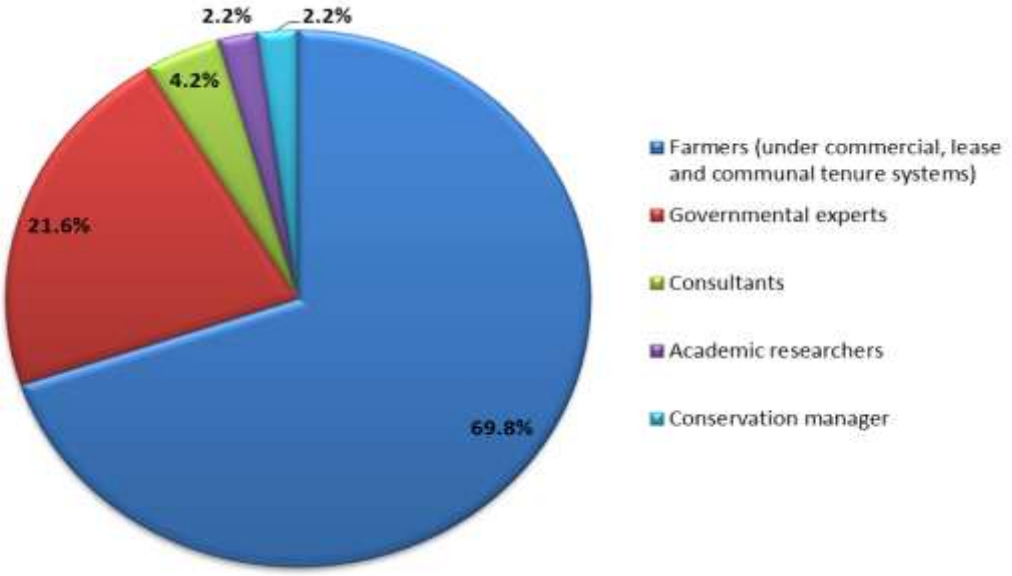

Figure 4.6: The proportion of each type of expertise within the multi-stakeholder platform.

4.3.2 IAPro step 2: Baseline evaluation of restoration and management actions and related indicators

The semi-structured interviews conducted with the 46 selected members of the MSP revealed that the rangeland management action most often applied within the study area boiled down to rotational grazing management with a four-camp fenced system whereby a camp is grazed by cattle at a stocking density of about 10 ha LSU^{-1} for two weeks and then rested for six weeks.

The rangeland restoration actions employed most often to mitigate rangeland degradation within the study area included (a) chemical aeroplane and manual bush control of unwanted thickening of woody shrubs and trees with the arboricide called Molopo CC (the active ingredient in the arboricide being Tebuthiuron) (Figure A8, in Appendix), and (b) re-vegetation by ripping the soil and reseeding with indigenous grass species, mainly in areas where the woody plants have been removed by way of mechanical (e.g. big machinery) or chemical methods (Figure A10, in Appendix).

Controlled sites were identified on the same rangelands where no mitigation actions have been applied. These included areas that are still bush-thickened (Figure A7, in Appendix) or where no rotational grazing (Figure A12, in Appendix) is implemented. The control sites are normally characterised by low rangeland productivity in terms of grazing.

During the interviews, a total of 31 site-specific ecological and socio-economic indicators (criteria) used by the local SHs to assess the effectiveness of the different applied restoration and management actions were mentioned (Table 4.2).

Table 4.2: The site-specific indicators with an explanation of each and the number of SHs (n=46) who mentioned each indicator during the interviews to assess the effectiveness of different restoration and management actions in the Molopo region of the North-West Province.

Indicator	Explanation of indicator	Number of SHs that mentioned indicator
Animal condition & value	Animal condition and quality, such as weaning mass and breed quality (including references to market value as affected by animal condition), as well as reproductive health of the animals	34
Efficacy & recurrence	The efficacy of the action to kill the target woody species causing the thickening, the amount of re-sprouting and extent of new establishment: This also includes any references to the need for retreatment either with the same or a new method.	34
Total costs & availability	All the costs (e.g. materials and labour) as well as the availability of the materials and labour for a certain management or restoration action	33
Grass abundance	The abundance of grasses regardless of the forage value: It includes attributes such as grass density, cover and frequency. It also includes references to the re-growth (from existing and new establishment) and recovery of the grass layer.	30
Forage value	This includes any references by an SH highlighting the palatability and forage value of a grass species. This is different from forage production in that it highlights species and type instead of quantity.	29
Forage production	Grass biomass production for current and future use: Note that the SHs used the term biomass and grass production interchangeably.	27
Specificity	Specificity of a bush control method as to whether non-target species such as grass and desirable woody plants are killed inadvertently.	23
Grazing capacity	Grazing capacity of the veld or the number of animals that can be supported on a rangeland without damaging the veld	21
Risks of failure or loss	The probability of suffering a loss that ultimately has a financial impact on the farming business, either because a management or restoration action fails or because of a catastrophe such as veld fire or drought	19
Biodiversity of plants	Biodiversity of plants across all functional groups regardless of forage value and any plant species composition, whether general or only grasses	18
Grass vigour	The vigour of grass growth such as basal cover (tuft size), seed production and root growth	17
Income & profit	This refers to any mentioning of income and/or profit from the farming business and includes references to technical metrics on returns such as ROAM (return on asset managed) and ROI (return on investment).	14
Speed	The amount of time it takes for a management or restoration action to manifest its effect. This also includes references to the speed of recovery of a degraded area.	14
Ground cover	Any references to ground cover (non-living cover on soil) and bare, open-ground patches.	13
Simplicity	The simplicity of a mitigation action, ease of use of equipment or technique and the need for specialised training or equipment	11

Indicator	Explanation of indicator	Number of SHs that mentioned indicator
Soil condition	Soil condition or quality such as soil fertility, organic matter content, soil stability and structure: This also includes the soil's ability to retain water and nutrients.	10
Animal protection & control	The ability to prevent animal theft or predation, as well as the ability to contain or separate animals as needed during the grazing management system	8
Drought preparedness	The system's ability to be sustainable through a drought period in terms of its ability to still maintain animals	7
By-products & services	Extra ecosystem services or products that result from a management or restoration action, such as fire control, creation of job opportunities, woody plants for fuel or other income-generating material	6
Non-plant biodiversity	Diversity of fauna (animals)	6
Soil erosion	Soil erosion or a system's ability to conserve soil: SHs make use of different terms to describe the indicators. Soil erosion is highly related to soil condition, which is a good example of redundancy among the indicators.	6
Worms and parasites	Worms or parasites that may develop associated with a management or restoration action	6
Toxicity & hazard	Toxicity of a substance used, and health hazard associated with a management or restoration action	5
Personal wellbeing	Family and personal wellbeing (their happiness and standard of living); the ability to care for the family over the long term	4
Woody plant abundance	The abundance (as in cover and density) of woody shrubs and trees: When an SH referred to plant cover or density without making it clear whether it refers to herbaceous or woody plants, this was captured as being mentioned in both the indicators – woody plant abundance and grass abundance.	4
Insect activities	Activities of insects such as caterpillars, dung beetles, ants and termites	3
Naturalness	The naturalness of a management or restoration action: For example, should re-vegetation be implemented as a restoration action, does the land user make use of indigenous plant species? Or if shrub clearing needs to be done, does the land user bring in camels or goats that do not naturally occur in the region to utilise and suppress the thickening of woody plants?	3
Habitat diversity	Includes the structure of the plant community.	2
Sustainability	Long-term sustainability of a management or restoration action	2
Animal distribution	Wildlife distribution on the landscape	1
Cultural lifestyle	The value attached to traditional and cultural lifestyle of the land users	1

A short-listing of environmental and socio-economic indicators proposed by the SHs that have been interviewed, the availability of data and the collectability thereof resulted in a final set of only 11 indicators that were used for the evaluation of the three restoration and management actions mentioned in step 2 of the IAPro. The 11 indicators representing socio-economic implications and the four ecosystem services are outlined below.

Socio-economic indicators:

1. **Income and profit** - The income and profit the land user gains after the implementation of restoration and management actions to improve the rangeland production.

2. **Bush control effectiveness** – The effectiveness of the restoration or management action to control and/or remove high densities of unwanted woody plants.

3. **Risks** – Wild fires and droughts that are a risk to the land users in the Molopo study area. These factors are capable of putting an end to forage production and causing animal deaths, which could ultimately have a negative financial impact. Another risk includes a shortage of water in these semi-arid environments. The latter is mainly experienced during periods of drought.

4. **Costs (both material and labour)** – The costs and availability of the materials and labour to implement a specific management or restoration action, such as fencing, watering points, water pumps and arboricides for bush control.

Ecosystem services:

The suite of science-based common indicators aims to represent a well-balanced basket of ecosystem services (ES) which cover four broad catergories, i.e. provisioning, regulatory, supporting and cultural services with the main focus on key services in arid ecosystems (Buatista & Orr, 2011). Criteria and the suite of indicators, including potential metrics for their assessment, proposed by the IAPro are summarised in Table 4.3.

Table 4.3: Science-based common criteria and indicators (proxies) for the assessment of management and restoration actions to combat rangeland degradation in arid regions (source: Bautista & Orr, 2011).

Criteria		Indicators/ Proxies
Economy	Income, personal wealth	Site-specific
Provisioning services	Goods (food, fibre, timber, fuel wood)	Productivity/ Productivity value
Regulating & supporting services	Water and soil conservation	Plant cover and pattern/ Soil surface condition
Cultural services	Landscape and cultural heritage	Site-specific
Biodiversity		Diversity of vascular plants

Provisioning services

5. **Forage production and value** - The amount of dry matter biomass produced by grasses that are available for the grazing animal.

6. **Grazing capacity** - The land area (hectares) required by 1 large stock unit (LSU) to sustain itself and be productive per year without damaging the rangeland. It largely depends on the forage production and the palatibilty of the grasses.

7. **Animal condition and value** - Meat quality, reproductive health and weaning mass of the animal. It stands in direct relation to the monetary value of an animal.

Regulating services

8. **Grass abundance** - The total grass cover and density of grasses (both new establishment and new growth of existing individuals), regardless of the forage value, regulating ecosystem integrity.

9. **Woody plant abundance** - The occurance and density of woody plants regulating ecosystem integrity.

Supporting services

10. **Soil condition and conservation** - Soil properties such as organic matter and structural stability that would support soil fertility. This also includes the soil's ability to retain water and other resources.

Biodiversity

11. **Biodiversity** – The number of different grass and other plant species found in a specific area where a restoration or managment action has been applied. Grass diversity was used as a proxy to quantify overall plant diversity.

4.3.3 Step 3: Indicator weighting exercise

4.3.3.1 Step 3a: Weighting of final set of indicators

During the first iteration of assigning a weight value to each indicator for action evaluation, the indicator "forage production and value" was regarded most important, followed by "grazing capacity" as the second most important indicator (Figure 4.7). The indicator "income and profit" was ranked third most important, followed by "animal condition and value" and "grass abundance" in the fourth and fifth position respectively. The indicator "soil condition and conservation" was ranked sixth and "biodiversity" seventh, which indicated that livestock farmers are more interested in perennial palatable grasses than different types of woody plants. "Total cost for material and labour" was ranked eighth, the "abundance of woody species" ninth, with "risks" being the least important indicator for evaluating management and restoration impacts (Figure 4.7).

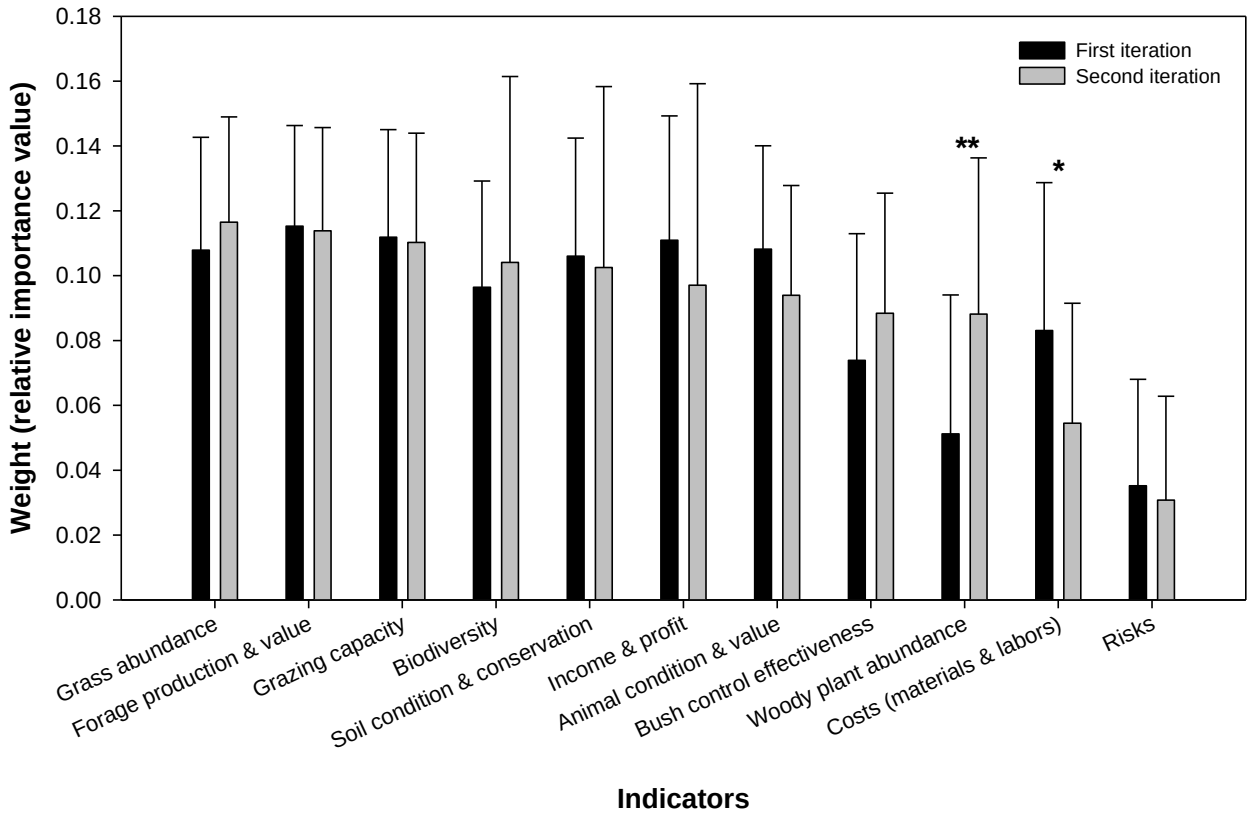

Figure 4.7: Relative importance values for the 11 indicators averaged over individual SH perceptions before (first iteration) and after (second iteration) group discussions: The sequence of indicators are according to the weight values of the second iteration ranked from the most important indicator on the left- to the least important indicator on the right-hand side of the graph. Bars represent means (±SD), and significant differences are indicated by asterisks at *$p < 0.05$ and **$p < 0.01$ (Mann-Whitney-Wilcoxon test).

4.3.3.2 Step 3b: Discussing individual and collective results and re-assessing the indicators

The second iteration of indicator weightings following group discussions within the multi-stakeholder platform revealed some differences in the relative importance (ranking) of indicators compared to the first iteration (Figure 4.7). SHs rated the indicators “grass abundance”, “bush control effectiveness”, “biodiversity” and “woody plant abundance” higher in the second ranking exercise, with the latter having a significantly higher mean weight (Figure 4.7). “Costs” were ranked significantly lower in the second iteration, with the same applying to “Income and profit” (ranked sixth) and “animal condition” (ranked seventh).

4.3.4 IAPro step 4: Quantification of indicators

Following the IAPro, the sampling sites for the quantitative assessment of indicators had to be on the same land that belonged to the members of the multi-stakeholder platform (MSP) who took part in the ranking procedure and who had implemented a management or restoration action to combat rangeland degradation. A total of 50 vegetation sampling sites were selected (Figure 4.3). The total number of sampling sites was located on livestock and game farms, i.e. seven commercial- (as they were the farmers who are spending the most time and money to increase rangeland productivity), three lease-, two communal- and two commercial game farms (Figure 4.3).

Biophysical indicators

Sampling sites with an application of rotational grazing, chemical control and re-vegetation showed a significantly higher mean forage production ($p < 0.05$) and grazing capacity ($p < 0.05$) compared to bush-thickened sites (Table 4.4). No rotational grazing had a significantly higher grazing capacity ($p < 0.05$) compared to bush-thickened sites but a significantly lower ($p < 0.05$) grazing capacity compared to the restoration (i.e. re-vegetation and chemical control) and management (i.e. rotational grazing) actions (Table 4.4).

In re-vegetated habitats, a significantly higher grazing capacity (5.8± 1.8 ha LSU^{-1}) ($p < 0.05$) was found compared to no rotational grazing and bush-thickened sites (Table 4.4), largely due to the high forage production (2120.1± 730.2 kg ha^{-1}) and high grazing value of grass species present. The second highest average grazing capacity (7.1± 1.6 ha LSU^{-1}) was found under rotational grazing due to the high abundance of perennial, palatable climax grass species (mainly *Schmidtia pappophoroides*) and second highest average forage production (2066.3± 336.9 kg ha^{-1}) (Table 4.4). In chemically controlled sites, a high average frequency of less palatable, perennial to weak perennial (pioneer-subclimax) grass species was found, hence the lower grazing capacity (10.2± 2.7 ha LSU^{-1}), compared to the re-vegetated and rotational grazed sites (Table 4.4). No rotational grazing showed a low forage production of 988.4 kg ha^{-1} and a significant lower ($p < 0.05$) grazing capacity of 35.7± 8.9 ha LSU^{-1}, compared to the re-vegetated (5.8± 1.8 ha LSU^{-1}), rotational grazed (7.1± 1.6 ha LSU^{-1}) and chemically controlled sites (10.2± 2.7 ha LSU^{-1}).

Bush thickening had a detrimental effect on forage production at only 410.8± 197.3 kg ha^{-1} when compared to rotational grazing with 2066.3± 336.9 kg ha^{-1} (Table 4.4). The mean grazing capacity of these sites was found to be 93.8± 48.7 ha LSU^{-1}. This is significantly lower ($p < 0.05$) compared to the sites where re-vegetation, rotational grazing management and chemical control actions were implemented (Table 4.4).

There were no significant differences ($p < 0.05$) in grass diversity (Shannon-Weiner diversity index) between the actions (Table 4.4). Grass abundances differed significantly when all the restoration and management actions and bush-thickened sites are compared (Table 4.4). Under no rotational grazing, the grass abundance was the highest with a mean value of 18.8± 2.6 plants m^{2-1}. Bush-thickened areas had a significant lower mean grass abundance (7.8± 3.5 plants $m^{2\ -1}$) ($p < 0.05$) (Table 4.4) compared to re-vegetated, rotational grazed and chemically controlled sites. In areas that showed a high bush thickening with high bush densities that have been controlled with chemicals, the grass abundance was about 14± 5.4 plants $m^{2\ -1}$, more or less the same as for rotational grazing (14.6± 4.9 plants $m^{2\ -1}$) (Table 4.4). Under the re-vegetation restoration action, a significantly higher grass abundance was found (15.1± 3.8 grass plants $m^{2\ -1}$) ($p < 0.05$) compared to the bush-thickened sites (Table 4.4).

In sites where rotational grazing was applied, an average of 264.5± 92.3 TE ha^{-1} (Table 4.4) was found, compared to the significantly lower ($p < 0.05$) woody density in the re-vegetated sites with only 44.4± 13.8 TE ha^{-1}. It seemed that chemical control was quite effective in reducing the average TE ha^{-1} to 237.1± 83.6, compared to an average of 1393.4± 453.2 TE ha^{-1} in the bush-thickened sites where no control was applied (Table 4.4). The average woody density in communal land-use systems (no rotational grazing) was only significantly lower (433.6± 26.1 TE ha^{-1}) ($p < 0.05$) in the bush-thickened sites (Table 4.4).

No significant differences ($p < 0.05$) were found within the percentage soil organic carbon between the sites where different restoration and mitigation actions were applied and the bush-thickened sites (Table 4.4). Although not significant, it was found that re-vegetated areas had the highest percentage organic carbon in the soil at 0.30± 0.09%, and bush-thickened sites (0.22± 0.07%) and chemically controlled sites (0.22± 0.04%) the lowest percentage soil organic carbon (Table 4.4).

Socio-economic indicators

It is important to note that it was not possible to collect sufficient quantitative data in order to test the validity of the socio-economic indicators (e.g. “income and profit”).

From the qualitative results, a significant ($p < 0.05$) positive influence was found for the indicators such as “income and profit” and the preparedness for “risks” following on the implementation of restoration and management actions when compared to bush-thickened sites (Table 4.4). However, there were significantly ($p < 0.05$) higher costs involved if actions such as re-vegetation (after shrub and tree clearing and then re-sowing), rotational grazing (erecting of fences and establishing better water reticulation) and chemical control (expensive chemicals and the application thereof) have been implemented. Animal condition was significantly ($p < 0.05$) higher in re-vegetated habitats and where rotational grazing management has been applied when compared to the bush-thickened areas and where no rotational grazing management has been implemented (Table 4.4). Chemical control of woody shrubs and trees had the most significant positive effect ($p < 0.05$) regarding the effectiveness of controlling woody plants when compared to the other restoration and management actions (Table 4.4).

Table 4.4: Effect of management and restoration actions when compared to bush-thickened sites on all parameters related to the final set of identified indicators used for action evaluation: Means (±SD) with different lower-case letters in a row indicate a significant difference at $p < 0.05$ (ANOVA with post-hoc Tukey's HSD test.

	Rotational grazing	No rotational grazing	Chemical control	Re-vegetation	Bush-thickened sites
Forage production (kg ha^{-1})	2066.3±336.9^{a}	988.4±540.4^{a,b}	1631.1±333.3^{a}	2120.1±730.2^{a}	410.8±197.3^{b}
Grazing capacity (ha LSU^{-1})	7.1±1.6^{a}	35.7±8.9^{b}	10.2±2.7^{a}	5.8±1.8^{a}	93.8±48.7^{c}
Grass diversity (Shannon-Weiner Index)	1.16±0.46^{a}	1.26±0.29^{a}	1.32±0.54^{a}	1.11±0.49^{a}	1.47±0.28^{a}
Grass abundance (plants m$^{2-1}$)	14.6±4.9^{a}	18.8±2.6^{a}	14.0±5.4^{a}	15.1±3.8^{a,b}	7.8±3.5^{b}
Woody density (TE ha^{-1})	264.5±92.3^{b}	433.6±26.1^{b}	237.1±83.6^{b}	44.4±13.8^{a}	1393.4±453.2^{c}
Soil organic carbon	0.25±0.09^{a}	0.28±0.06^{a}	0.22±0.04^{a}	0.30±0.09^{a}	0.22±0.07^{a}
Income and profit (Rank)†††	1.9±0.9^{a}	3.7±1.0^{b}	2.4±1.0^{a}	2.1±1.2^{a}	4.4±1.3^{b}
Animal condition (Rank)* **	1.8±0.8^{a}	3.7±1.0^{c}	2.8±1.0^{b}	1.3±0.8^{a}	4.8±0.6^{d}
Bush control effectiveness (Rank)* **	2.3±0.8^{b}	3.5±1.0^{c}	1.5±0.8^{a}	2.5±0.9^{b}	4.8±0.5^{d}
Costs for material and labour (Rank)* **	2.1±1.0^{a}	4.1±0.5^{b}	1.9±1.0^{a}	1.9±0.9^{a}	4.7±0.7^{b}
Risks (Rank)* **	2.2±1.1^{a}	3.6±1.0^{b}	2.6±1.1^{a}	1.9±1.1^{a}	4.5±1.3^{b}

††† The effect of the mitigation and restoration action on the indicator as assessed by the SHs (1 = having the most positive effect on the indicator, 5 = having no or the least positive effect on the indicator)

4.3.5 IAPro step 5: Integrating quantitative assessment and indicator data with SH perspectives

The MCDA revealed that in pair-wise comparisons, re-vegetation (RV) was the best mitigation action, outranking rotational grazed (RGM), chemically controlled (CC), no rotational grazed (NRG) and bush-thickened sites (BTS). Rotational grazing management was the second most effective action to combat rangeland degradation compared to the rest of the actions (Figure 4.8).

The higher ranked indicators, such as “forage production” and “grazing capacity”, were the reason for the outranking relationship of the three actions (RGM, RV and CC) when compared to the no rotational grazing and bush-thickened sites (Figure 4.8). “Forage production” and “grazing capacity” were also the second and third most important indicators ranked by the MSP as explained earlier (compare section 4.3.3.2).

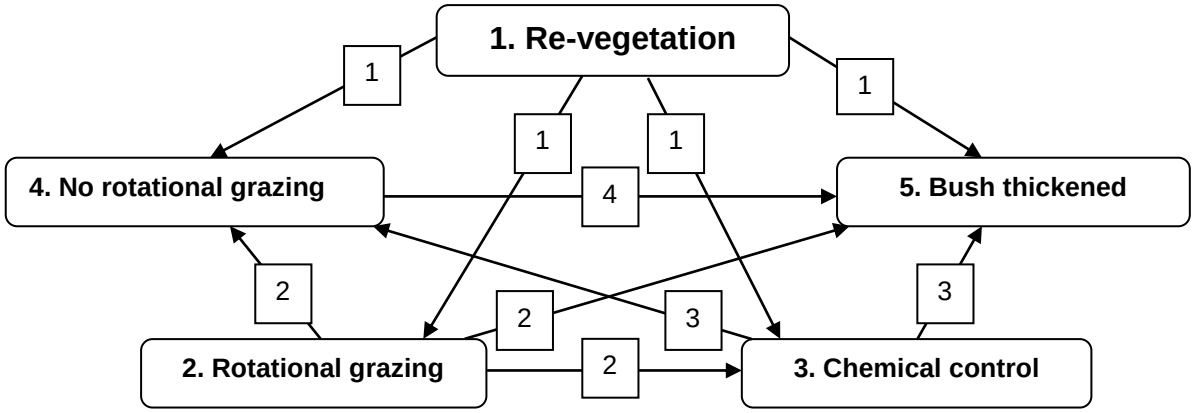

Figure 4.8: Outranking relationships of the MCDA performed for the five different actions (direction of arrows indicates an outranking relation with respect to the indicators between the actions): Numbers indicate the ranking order of actions (1 was the most effective method to combat rangeland degradation, with 5 being the least effective).

Ten out of the 11 indicators were found to be indifferent (i.e. no difference regarding the indicator between the two actions) among rotational grazing management and re-vegetation (Table 4.5). There is a strict preference for soil organic carbon in the re-vegetated sites, making re-vegetation the most effective action to combat rangeland degradation and improve rangeland conditions if soil organic carbon is used to compare the mitigation actions (Figure 4.5).

Table 4.5: Output of the MCDA comparing the response of the 11 indicators to re-vegetation as opposed to rotational grazing management (RV = re-vegetation; RGM = rotational grazing management).

Re-vegetation vs. rotational grazing			
Indifferent	**Weak preference for RV**	**Strict preference for RV**	**Strict preference for RGM**
• Grass abundance • Forage production • Grazing capacity • Woody plant abundance • Animal condition • Bush control effectiveness • Costs for materials • Risks • Income and profit • Grass diversity		• Soil organic carbon	

Rotational grazing management showed to be more successful in combating rangeland degradation than chemical control (Figure 4.6). There was only a weak preference in favour of rotational grazing management over chemical control, regarding the indicators of “forage production”, “soil organic carbon” and “animal condition” (Table 4.6). A strict preference in favour of chemical control regarding “bush control effectiveness” was, however, revealed by the MCDA analysis, because where the chemical control mitigation action had been applied, the second lowest woody plant densities have been recorded (Table 4.4).

Table 4.6: Output of the MCDA comparing the response of the 11 indicators to rotational grazing management as opposed to chemical control (RGM = rotational grazing management; CC = chemical control).

Rotational grazing vs. chemical control			
Indifferent	**Weak preference for RGM**	**Strict preference for RGM**	**Strict preference for CC**
• Grass abundance • Grazing capacity • Woody plant abundance • Costs for materials • Risks • Income and profit • Grass diversity	• Forage production • Soil organic carbon • Animal condition		• Bush control effectiveness

By comparing rotational grazing with no rotational grazing (i.e. continuous grazing), it was found that these two actions were indifferent with respect to the three indicators of “soil organic carbon”, “woody plant abundance” and “grass abundance” (Table

4.7). Strict preference was given to rotational grazing management regarding a higher “forage production”, which also resulted in the improvement of indicators such as “animal condition”, thereby increasing the land user’s “income and profit”. Strict preference for “grass diversity” was afforded to no rotational grazing management over rotational grazing (Table 4.7).

Table 4.7: Output of the MCDA comparing the response of the 11 indicators to rotational grazing management as opposed to no rotational grazing (RGM = rotational grazing management; NRG = no rotational grazing).

Rotational grazing vs. no rotational grazing			
Indifferent	**Weak preference for RGM**	**Strict preference for RGM**	**Strict preference for NRG**
• Soil organic carbon • Woody plant abundance • Grass abundance	• Bush control effectiveness • Grazing capacity	• Forage production • Animal condition • Costs for materials • Risks • Income and profit	• Grass diversity

Bush-thickened sites were only indifferent regarding “soil organic carbon” from the other actions, i.e. rotational grazing, re-vegetation and chemical control (Table 4.8). Strict preferences for the remaining 10 indicators have been obtained for the three restorations and management actions (i.e. RGM, RV and CC) over the bush-thickened sites. Due to the strict preference of indicators for rotational grazing management, re-vegetation and chemical control when compared to the bush-thickened sites, these restoration and management actions can, therefore, be regarded as being more effective when combating rangeland degradation and improving the condition of the rangeland (Figure 4.8).

Table 4.8: Output of the MCDA comparing the response of the 11 indicators to rotational grazing management (rotational grazing can be substituted with either re-vegetation or chemical control; the same results will be obtained) as opposed to bush-thickened sites (RGM = rotational grazing management; BTS = bush-thickened sites).

Rotational grazing/ re-vegetation/ chemical control vs. bush-thickened sites

Indifferent	Weak preference for RGM/ RV/ CC	Strict preference for RGM/ RV/ CC	Strict preference for BTS
• Organic carbon		• Grass abundance • Forage production • Grazing capacity • Woody plant abundance • Animal condition • Bush control effectiveness • Costs for materials • Risks • Income and profit • Grass diversity	

4.3.6 IAPro step 6: Collective integrative assessment

When the SHs were asked to rate the actions, the majority rated rotational grazing (78.6%) as a very good to excellent choice, whereas no rotational grazing was rated overall as a bad to very bad choice (92.9%, Table 4.9). Compared to the first assessment conducted in IAPro step 2 (the pre-interactive assessment, Table 4.9), the rating for rotational grazing remained basically the same. Chemical control was rated as a very good to excellent choice in both assessments (pre- and post-integrative assessments). During the post-integrative assessment, re-vegetation was rated by more SHs as a very good to excellent choice with no respondents rating it as a bad to very bad choice, compared to the 24.2% who rated it as a bad to very bad choice during the pre-integrative assessment (Table 4.9).

Table 4.9: SH ratings on the implementation of the various restoration and management actions as a good or bad choice by making use of a Likert scale (pre-/post-integrative assessment (% of responses)).

	Rotational grazing	No rotational grazing	Chemical control	Re-vegetation
Very good to excellent choice (%)	80 / 78.6	4 / 7.1	70.7 / 50	51.6 / 64.3
Moderate choice (%)	20 / 21.4	8 / 0	27.3 / 42.9	24.2 / 35.7
Bad to very bad choice (%)	0 / 0	88 / 92.9	3 / 7.1	24.2 / 0

4.4 Discussion

4.4.1 IAPro step 1 and 2: Stakeholder identification and baseline evaluation of restoration and management actions and related indicators

The successful undertaking of Step 1 can be ascribed to the following reasons: (a) Forty-six SHs could be identified to constitute the MSP. This is regarded as an appropriate sample size for this type of participatory research. However, the more SHs involved in the project the better, as this reduces bias and increases the chances of a more representative and comprehensive group of SHs (Bautista & Orr, 2011). The process of chain referrals was terminated when the referrals (as many as possible) were starting to become duplicative. (b) A variety of local SHs from various backgrounds (i.e. farmers, government extension officers from DARD, consultants, conservation managers and rangeland ecologists) could be identified to participate as a MSP in the study through chain referrals. (c) Land users and government experts, who have been farming or working in the study area for many years and who have good indigenous knowledge regarding locally applied restoration actions and rangeland degradation, were able to participate in the project. Therefore, SHs that made up the MSP were primarily represented by farmers and governmental experts. (d) SHs were able to contribute to the process of identifying locally applied restoration and management actions and site-specific indicators selected for actions evaluated in the Molopo rangelands. (e) The overall response of the members from the MSP was positive, with SHs willing to participate in and learn from the integrated assessment approach.

The results of step 2 were also very promising, as the type of participatory assessment helped to identify best and most sustainable actions to mitigate rangeland degradation. SH feedback helped in the identification of relevant site-specific indicators and could be used in the quantification of the indicators, particularly those that are of practical value to the land users.

Consequently, the main outcome of step 2, which included the identification of indicators for assessing actions to combat desertification and rangeland degradation, could be attained. A total of 31 ecological and socio-economic indicators were mentioned by the SHs, indicating a wealth of indigenous knowledge and information

available among the SHs in the regions of the Kalahari, such as Mier (Northern Cape Province) and Molopo (North-West Province) (Reed *et al*., 2008; Dreber *et al*., 2013).

A comparison between the Molopo and Mier farming communities (see Dreber *et al*., 2013), where the IAPro was also implemented, also revealed that similar problems relating to rangeland degradation are being faced here. It was documented that in both these study areas, an abundance of site-specific indicators (i.e. 31 for the Molopo and 30 for Mier) are used by the local land users to evaluate restoration and management actions based on biophysical and socio-economic indicators. Furthermore, the indicators identified in both the Molopo and Mier farming communities were very similar, including aspects such as grazing capacity, forage production, grass abundance, woody plant abundance, animal condition, soil condition, biodiversity and economic costs (Dreber *et al*., 2013).

Due to the many indicators reported in this study as conducted in the Molopo, a complex variety of land-user perspectives have been addressed and, consequently, the likelihood of the set of indicators being incomplete has been greatly reduced (Dreber *et al*., 2013). However, it was quite a challenge in the integrative process to identify the most suitable set of indicators due to the highly varying types of indicators suggested by all the SHs. Nevertheless, a final set of 11 indicators could be identified, short-listing criteria that were similar including popularity and collectability. Site-specific indicators identified by local SHs were combined with expert selected common indicators based on the ecosystem-services approach since this represents overall functioning of dryland ecosystems and ensured that the indicators could be quantified. A short-listing of the indicators (i.e. final set of only 11 indicators) was necessary to avoid too much detail and duplication at the expense of a final overview (Sommer, 2011).

The second main outcome of step 2 was to identify locally applied restoration and management actions that are used to combat rangeland degradation and desertification. This outcome was reached when SHs identified three main restoration and management actions commonly applied by land users to combat rangeland degradation and to improve the agricultural potential of these habitats. Most commercial tenure systems are based on a multi-camp approach whereby a rotational grazing management system is implemented. The rotational grazing

system includes a resting period, which allows for a recovery period between grazing periods with a view to better vegetation production over the long term (O'Connor *et al.*, 2010). As a result of the better vegetation production commonly found in correctly stocked, rotational grazing systems, this is the most commonly applied restoration action implemented by land users in the Molopo study area.

In highly bush-thickened areas, mitigation actions must also be implemented to restore the agricultural production potential of the rangelands and to increase the forage production of the land. To control bush thickening, chemical control technologies, using the arboricide Molopo CC, are implemented. This action has been identified as being very effective to control high woody plant densities in the Molopo region. As a result, chemical control is commonly applied in the study area mainly due to the effectiveness of the arboricide and the resulting increase in grass phytomass productions.

In highly bush-thickened areas that are chemically controlled and characterised by annual, less palatable grass species, re-vegetation practices are often applied. The latter is accomplished by ripping the soil and re-seeding with indigenous climax, more palatable grass species in areas where the woody plants have already been removed through mechanical methods. However, re-vegetation is an expensive action to apply and is not always very successful, especially if a low rainfall season follows the re-seeding activities. Consequently, re-vegetation is a less commonly applied action in the Molopo region. The advantage of re-vegetation as a restoration practice is, however, that the perennial grasses used in the re-seeding process provide the much needed ground cover and prevent soil erosion. The ecological disadvantage of this practice is that all the woody plants are removed before re-seeding is done, creating open grasslands which result in less soil nutrients and organic material provided by the dead woody plants as well as a low woody and grass plant diversity being established over time. Livestock farmers, however, seem to not regard these ecological disadvantages as negative, as they do not need a high diversity of grasses because they are mostly interested in the high forage production which is provided by the climax, perennial, palatable and large tufted grasses that are re-seeded (O'Connor *et al*., 2010).

Prescribed fires have been suggested as being cheaper than chemical bush control actions (e.g. Trollope, 1992; Buss & Nuppenau, 2003). However, SHs in the Molopo do not prefer to make use of fire as a restoration method due to the heightened risks involved when fire is used as a management tool. According to the SHs, the land that is burned would need to be rested for at least two years by either withdrawal of all grazers or re-seeding methods. Another reason cited is that heavily thickened sites do not have enough fuel loads from grasses to ensure a fire hot enough to kill the woody plants. Managing prescribed fires may also require extensive skills and experience that are not available to all the land users (Reed *et al*., 2007).

An integration of the local knowledge regarding restoration and management actions with ecological scientific knowledge (science-based common indicators) to combat rangeland degradation would yield more information on the dynamics of ecosystems and rangelands and the response to management than by only making use of ecological methods, as has also been found in other studies (e.g. Sheuyange *et al*., 2005; Fabricius *et al*., 2006; Reed *et al*., 2007; Stringer & Reed, 2007; Verlinden & Kruger, 2007; Oba *et al*., 2008[a]).

4.4.2 Step 3: Indicator weighting exercise

4.4.2.1 Step 3a: Weighting of final set of indicators

The overall pattern found with the ranking of the final set of 11 indicators was that farmers mainly manage their rangelands in order to achieve the highest “grass forage production” with a good “grazing capacity” (ranked 1st and 2nd respectively during the first iteration) and not to maintain a high woody or “grass diversity” (ranked 7th). Similar results were found by O’Connor *et al*. (2010). It seems that the lower ranking of the “grass diversity” indicator can be ascribed to the fact that cattle farmers dominated the MSP, and they are more interested in a high grass forage production. These results show how the outcome of the PRACTICE IAPro approach as a whole can be determined by the dominant SH group identified in step 1.

The indicator “high woody plant abundance” was mentioned by the SHs during the interviews in step 2 as having a highly negative influence on grass growth (second lowest ranking in importance), the same identification as per farmers from the Mier community (Dreber *et al*., 2013). The low ranking of this indicator, i.e. being less important, may be ascribed to the fact that the SHs misunderstood what was meant

by “woody plant abundance” and maybe perceived it as being an indication of high woody plant densities, which are less important. This emphasises the importance of conveying the correct knowledge to the SHs and to make sure that the information is correctly understood.

Degradation indicators relating to the density of certain shrub or tree species are commonly used in other parts of the Kalahari, such as the northern parts of the Northern Cape Province in South Africa and in southern Botswana (Reed *et al.* 2008). However, in a study conducted by Jacobs (2000) in the Kalahari Thornveld where local land users were interviewed, it was found that the land users did not consider bush thickening to be an environmental problem, because they are too poor to have many cattle that are dependent upon the higher forage production for grazing. They mainly keep small stock and donkeys which are able to utilise the woody species by browsing.

Another reason for not regarding woody thickening as a problem is that the woody species (such as *Acacia erioloba* and *Boscia albitrunca*) have a better survival rate during droughts and can then be used as fodder by the browsing animals. Trees also provide shade (Jacobs, 2000; Seymour, 2008), especially in the dry, hot summer season. These factors may, therefore, account for the reason why certain farmers ranked the “woody plant abundance” indicator much lower when compared to the other indicators in the present study.

The “risks” indicator, which includes droughts or wild-fires, can have a huge impact on rangeland and animal condition (Scholes *et al*., 2002; Sankaran *et al.,* 2004, 2005; Bond *et al.,* 2005; Sankaran & Anderson, 2009; Fensham *et al.,* 2009; Buitenwerf *et al*., 2011; Joubert *et al*., 2012). However, this indicator is ranked as least important, as risks are not that common in this region (Trollope, n.d. *in* De Klerk, 2004; Scholes, 2009; Joubert *et al*., 2012).

As mentioned above, difficulties observed during the indicator ranking exercise related to the fact that certain SHs did not quite understand or had different perceptions of what was meant by some of the terms and indicators used, e.g. “biodiversity” and “rangeland degradation”. Similar issues have been identified in studies conducted by Reed *et al*. (2008) and Dreber *et al*. (2013) who have all worked in the Kalahari region. These indicators had to be clearly explained in

participants' native language (Afrikaans or Setswana) before ranking could take place. This problem also became evident when land users from different land tenure systems and cultural backgrounds were grouped together, especially regarding the ranking of the biophysical and socio-economic indicators, as the contexts differed quite significantly between, for example, the commercial and communal management systems (Reed *et al*., 2006; Von Maltitz, 2009, Dreber *et al*., 2013).

4.4.2.2 Step 3b: Discussing individual and collective results and re-assessing the indicators

The "grass abundance" indicator was ranked highest during the second and final iteration (Figure 4.7). It is assumed that this indicator is used to assess whether a restoration or management practice had been effective and whether the frequency of bare patches in bush-thickened areas had been reduced, providing the necessary soil cover.

Likewise, the indicator "woody plant abundance" was ranked significantly higher in the second ranking exercise (Figure 4.7). It is assumed that the SHs might have realised that higher woody plant densities in bush-thickened sites will have a high negative effect on the forage production and rangeland condition, especially after the quantitative results from the biophysical results (as derived during step 4) have been presented to the farmers. Conversely, the "costs" indicator was ranked significantly lower in the second iteration (Figure 4.7), maybe due to the realisation of farmers that it is worth spending money on restoration and management practices in order to retain or improve rangeland conditions, thereby increasing their profit and income.

It is not clear why "income and profit" and "animal condition" were also ranked slightly lower during the second iteration (Figure 4.7). As mentioned by the SHs during the workshops, if the first five indicators (i.e. "grass abundance", "forage production", "grazing capacity", "biodiversity" and "soil condition") are in a good condition and in place, the animals will also be in a good condition, contributing to a higher income and profit. The "bush control effectiveness" and "woody plant abundance" indicators were ranked higher during the second iteration, presumably because farmers became more aware of this increasing problem during the workshops and group discussions held between farmers who have seen and heard about the negative effects high woody plant densities can have, i.e. decreasing

forage production and grazing capacity. The occurrence of “risks” remained the lowest indicator in both iterations (Figure 4.7). SHs’ reasons for this is that they are more prepared for wild fires with the aid of the necessary fire-fighting equipment and preparatory actions implemented before the onset of the wildfire season, especially in the Molopo region. The SHs also reason that if the mitigation actions to combat rangeland degradation are implemented properly, they will have enough grazing and browsing material available to carry them through a period of drought.

The final result and outcome for this step (3b) of the IAPro was successfully reached with a final set of 11 indicators and a weight value assigned to each indicator to be applied in the evaluation (step 5 of the IAPro) of the restoration or management actions. Step 3 also represented the first opportunity within the IAPro for knowledge exchange/ social learning and the integration of scientific and local knowledge (Rojo *et al*., 2012). This participatory approach links science to society by combining local knowledge with ecological expertise for improved decision making and social welfare. It stimulates SHs to think about which indicators to use when making more informed decisions in the implementation of restoration and management actions in the future.

SHs provided information on their preferences of indicators they themselves have identified but also those that have been identified as a result of sharing the views of other SHs. Consequently, SHs’ original views may have been affected by the exchange of knowledge and perceptions during the workshops and group discussions between SHs. The two weighting exercises of the 11 indicators, group discussions during workshops and collective integrated assessments established a more diversified understanding of the environment of the SHs, which is based on local indigenous and scientific expertise (Fraser *et al*., 2006). This underlines the importance of social learning.

Although similar indicators were mentioned in studies conducted in other areas of the Kalahari (e.g. the Mier area in the Northern Cape Province), the weighting values ascribed to the indicators differed from the Molopo farming community, indicating that the indicators’ perceived importance is context-specific (Dreber *et al.*, 2013). The indicators “grass abundance”, “grass forage production”, “grazing capacity” and “biodiversity” were ranked the highest in the higher rainfall Molopo area, whereas the

“availability of water”, “animal condition”, “grazing capacity” and “personal factors” (wellbeing) were regarded as the most important indicators in the lower rainfall area of Mier. In the Mier farming community, water availability ultimately determines the application of better management strategies such as rotational grazing management.

According to the integrated assessment protocol, the ranking of indicators, subsequent group discussions and the re-ranking of indicators should ideally commence during a single meeting (Bautista & Orr, 2011). Unfortunately, due to the remote location of the sampling sites in combination with time- and budget constraints to complete the biophysical assessment (IAPro Step 4) at the different study sites in the Molopo rangelands, more than one year passed before the second iteration of the ranking exercise (IAPro Step 3b) could be completed.

In the year that passed between the two iterations, other factors (such as financial constraints or lower rainfall) could also have played a role for the change of mind about and differences in ranking the indicators. Yet another reason for a change in the ranking can be ascribed to the fact that not all the SHs took part in the first (step 3a) and second (step 3b) iteration and ranking. The vast distances SHs needed to travel to attend workshops and their other obligations may well have resulted in varying participation rates, which could be problematic when attempting to ensure a sound interpretation of the results. Naturally, the indicators considered to have the highest importance values by the SHs will ultimately determine the action that is indicated by the MCDA analysis as the most effective action to combat rangeland degradation and improve rangeland conditions.

The above-mentioned constraints have to be considered when analysing the output of step 3, which may also be of relevance regarding the outcomes of subsequent steps, particularly Step 5.

4.4.3 IAPro step 4: Quantification of indicators

Biophysical indicators

A total of six indicators were quantitatively evaluated. Reduced “grass forage production” and “grass abundance” are two indicators that had been identified by SHs and also supported by field assessments as the main reason for the shift towards lower grazing capacities. This is consistent with the indicators used by land users in other arid environments (e.g. Oba & Kaitira, 2006; Reed *et al.*, 2008). The “grass forage production” indicator was found to be higher compared to no rotational grazing and bush-thickened sites in all three restorations and the management actions (i.e. rotational grazing management, re-vegetation and chemical control) that have been assessed. It is evident that under the no rotational grazing management system, where no camps are used to allow resting between grazing periods with the intention to ensure recovery of the vegetation after having been utilised, overutilisation and land degradation are inevitable (Hardin, 1968; Everson & Hatch, 1999; Hoffman & Todd, 2000). From the study by Smit and Swart (1994), it is evident that the competition for soil moisture between grass and woody plants in high bush-thickened densities resulted in the suppressing of grass establishment and growth, as mentioned earlier. Therefore, a lower grass “forage production” and “grazing capacity” was found in the bush-thickened sites, compared to rotational grazing management, re-vegetated and chemically controlled habitats.

Although the SHs proposed “grass diversity” as an indicator to assess rangeland degradation, no significant differences could be found in “grass diversity” (Shannon-Weiner diversity index) amongst the various actions (Table 4.4). Similar results were found by Kgosikoma *et al*. (2012). Overall, the grass diversity (Shannon-Weiner diversity index) was rather low across all the sites (Table 4.4). It therefore seems that the vegetation tend to be well adapted to a low mean annual precipitation in these semi-arid Molopo regions and that a low species diversity can be expected for this area in comparison to regions with a higher precipitation (O’Brien, 1993; Shackleton, 2000; Scholes *et al*., 2002; McNeely, 2003).

In this study, the lowest diversity was recorded in re-vegetated areas, as the woody plants causing thickening and competing highly with the grasses have been removed before a mixture of perennial, palatable climax grasses (e.g. *Cenchrus ciliaris*) was

re-seeded in the cleared areas. Due to the high competitiveness of *Cenchrus ciliaris* regarding water, light and nutrient absorption, the re-vegetated areas were largely dominated by this palatable, climax grass species.

Only two woody plant-based indicators were regarded as important by the SHs, i.e. “woody plant abundance” and “bush control effectiveness”. High density stands of woody plants such as *Acacia mellifera, Dichrostachys cinerea* and *Terminalia sericea* were regarded by SHs as a huge problem, as these high woody plant densities cause a reduction in grass production, not only in the Molopo rangelands but also in other areas of the Kalahari in southern Africa (Skarpe, 1990[a]; Thomas, 2002; Kgosikoma *et al.*, 2012).

All three restoration and management actions evaluated by this study were highly effective in suppressing and mitigating high woody plant densities (see section 4.3.4). This result is not consistent with the indicators used by land users in other arid environments (Reed *et al*., 2008). These land users suggested a reduction in woody plant density as an indicator of rangeland degradation. However, the latter was not supported by the biophysical measurements carried out by Reed *et al*. (2008) in the southern parts of Botswana.

Only one soil-based indicator, i.e. “soil condition” was mentioned by the SHs to be important when assessing rangeland conditions. This encompasses factors such as soil organic carbon (SOC), which plays an important role in maintaining the soil’s physical, chemical and biological properties (Tongway & Ludwig, 2011). Bulk density, infiltration rate and water holding capacity of the soil are improved with a higher occurrence of soil organic carbon (Doerr *et al*., 1998; Knicker, 2007). However, the Kalahari sandy soils are typically deep, structure-less and low in soil organic matter (Bergström & Skarpe, 1985; Skarpe & Bergström, 1986; Dougill *et al*., 1998).

Based on the results of this study, no significant differences could be found regarding the percentage organic carbon between the three different restoration and management actions, as well as the bush-thickened sites (Table 4.4). Although not significant, it is important to note that bush-thickened areas and former bush-thickened areas that have been treated with chemicals had the lowest soil organic

carbon (Table 4.4). In other degraded dryland environments, a reduction in SOC has also been observed (e.g. Dougill *et al*., 1999; Hill & Schütt, 2000).

Socio-economic indicators

As previously mentioned, it should be noted that it was not possible to quantify popular socio-economic indicators (e.g. “income and profit” and “risks”) directly, the reasons being that these indicators did not specifically constitute the rangeland conditions or were regarded as confidential. Given the lack of available data for the socio-economic indicators, land users may find it difficult to employ many of these indicators. The qualitative ranking procedure that was selected to gain data on these socio-economic indicators proved to be too abstract for some of the SHs who participated, and as also suggested by Dreber *et al*. (2013), this needs to be addressed or adapted for the sake of conformability.

As explained above, according to the SHs, a higher grass forage production and grazing capacity gained after the implementation of restoration and management actions will result in a higher “income and profit” for the land user compared to bush-thickened sites. SHs ranked it to be much more cost effective to implement or manage a rotational grazing system if sustainably managed to improve income and profit than to implement re-vegetation actions (see section 4.3.4). Land users also argued that they are better prepared for “risks” (e.g. droughts and wild-fires), especially when implementing restoration and good management actions, as these result in better rangeland production and available fodder over the long term. Under continuous heavy grazing (no rotational grazing) and with an increase in bush thickening, animals in poorer conditions need to be sold or moved to areas with better forage production, which will also contribute to a lower income and profit for the land user.

Chemical control was ranked significantly less positive regarding the indicator “animal condition” if compared to re-vegetation and rotational grazing management, as the grass forage production was lower in the chemically controlled sites (Table 4.4). Chemical control may also have negative effects, such as the removal of woody plants that could have provided some shade for animals, fodder for the browsing animals and pods that could be utilised by both livestock and game, especially during the dry season (Seymour, 2008).

From the results obtained during the biophysical assessments (Step 4), the re-vegetation action was the most effective regarding the removal of shrubs and trees (Table 4.4). However, re-vegetation is ranked as having a less positive effect on "bush control effectiveness" by the SHs when considering the time and costs involved to implement re-vegetation actions. The high costs are mainly for clearing and removing all the woody plants with a bulldozer or by hand, after which re-seeding is done. Although also costly, chemical control by aeroplane seems to be much less time consuming. Far more risks also seem to be involved in the implementation of re-vegetation, as an average or above-average rainfall season is needed to ensure the successful establishment of the sown seed.

A shortcoming identified for step 4 is based on the preferential sampling approach that has been followed, since this often involved the subjective selection of sites (Dreber *et al*., 2013). It was sometimes quite difficult to identify enough sampling sites that were suitable for replications to reduce variability in the data resulting from heterogeneity due to the biophysical environment and differences in the design of restoration and management strategies applied. However, this type of site-selection approach provides a measure of control by the local SHs and, therefore, increases the overall acceptance in survey site selection (Van Rooyen, 1998).

4.4.4 IAPro step 5: Integrating quantitative assessment and indicator data with SH perspectives

Due to the fact that there were many indifferent indicators as far as rotational grazing management and re-vegetation are concerned, these actions were revealed in the MCDA analysis as being equally good as effective actions to combat rangeland degradation and to improve rangeland conditions. However, in order to find an action that would be the most effective, should all be compared with one another, a higher cut level (0.9) had to be used in the MCDA (see section 4.2.1, step 5). After increasing the cut level value, a clear distinction could be made regarding the action that was the most effective. It was found that re-vegetation was the most effective action to combat rangeland degradation and to improve rangeland conditions, with rotational grazing management as the second best, followed by chemical control.

The results from the MCDA analysis found that the no-mitigation action (bush thickening) and no rotational grazing were less effective in combating rangeland

degradation compared to rotational grazing management, re-vegetation and chemical control (Figure 4.8), mainly due to the very low “grass forage productions” and “grazing capacity” found in these actions. By comparing rotational grazing with no rotational grazing, it was found that these two actions were indifferent with respect to “woody plant abundance”. Woody plant densities of the no-rotational grazed sites in the communal managed sites were rather low, which is likely due to the anthropogenic over-utilisation of woody plants for construction material and burning fuel, especially near the villages where the assessment had been conducted (also observed by Nkambwe & Sekhwela, 2006).

As previously mentioned, strict preference for “risks” was given to rotational grazed, re-vegetated and chemically controlled sites over no rotational grazed and bush-thickened sites, as the land users said during the second workshop that they are better prepared for drought periods after resting certain parts of the rangelands that can be used as additional available fodder if needed.

It is clear that if a more sustainable rangeland management strategy, such as rotational grazing management, is to be applied in especially bush-thickened areas, the rangeland must first be cleared or the trees be thinned out to reduce the competition with palatable, climax and perennial grasses. The latter and choice of clearing method to control woody species will depend on the financial status of the land user, as well as the specific land-use objective. Re-vegetation actions, which mostly involve the total clearance of trees and shrubs, is a rather drastic intervention eliminating any competitive effects in favour of an increased phytomass production of grasses. However, it must be profitable particularly for commercial cattle farmers.

If hunting is practised on the farm, the open spaces created by the re-vegetation can also be favourable, as they increase the hunter’s line of sight. The open spaces can also have a positive aesthetical value for the eco-tourism sector. Furthermore, it is advised that the chemical control of the woody component in bush-thickened sites be carried out selectively, as the environment created will be more natural providing important key resources for browsing herbivores by the remaining trees and shrubs (be it either game or goats) whilst creating fertile islands for grass establishment (Schlesinger *et al*., 1990).

It should be noted that the MCDA analysis does not identify the absolute best action that can be applied, but rather provides relative information on the ranking of the different actions that have been evaluated as a function of the 11 indicators considered in step 3 of the IAPro (Bautista & Orr, 2011).

4.4.5 IAPro step 6: Collective integrative assessment

Step 6 targeted the integrated evaluation of the locally applied restoration and management actions to combat rangeland degradation by encouraging the re-evaluation of the actions according to the results from the MCDA analysis and previous discussions by SHs during the first workshop. By combining what has been learned by the SHs in previous steps (i.e. social learning and discussions at workshops) and the quantitative results obtained through the scientific assessment (i.e. biophysical data), the local SHs ought to be able to make more informed decisions regarding restoration and management actions on their land.

Rotational grazing was rated very high and seems to be an excellent choice to implement (Table 4.9), as is proven by the quantitative data from the biophysical assessments. This management action will also mitigate the occurrence of high woody plant densities as a high grass abundance and good grass forage production (the two indicators regarded as most important by the SHs, see section 4.3.3) are maintained. High grass abundances suppress woody seed germination and prevent shrub thickening.

Chemical control was rated by only 50% of the SHs as a good to excellent choice (Table 4.9). This rating can be higher if selective manual chemical control is applied in order to only remove certain unwanted woody plants, thereby maintaining valuable woody plant species that can provide shade for the animals and fodder (leaves and pods) during the winter months (Richter *et al*., 2001; Seymour, 2008).

If selective control is possible, chemical control of woody species may be regarded as a better choice to implement instead of re-vegetation, as the cost of the latter is likely to be much higher. It must, however, be kept in mind that the re-vegetation action contributes to a higher “grass forage production”, “grass abundance” and “grazing capacity” (Table 4.4), the three indicators assigned the highest weight values in step 3b during the second iteration and preferred by cattle farmers focusing

on grass production (Figure 4.7). As many sites chosen for this study fell on cattle farms, it may be the reason why re-vegetation was regarded as a better choice to implement than chemical control.

The implementation of the restoration and management actions will ultimately depend on the costs of each action and financial ability of the land user. Another constraint in the implementation of a specific restoration or management action is the aspect of time and how long it will take to attain the preferred results. It will, for example, take longer to implement re-vegetation or rotational grazing actions to achieve a higher grass production, compared to any chemical control action where the results can be seen much quicker, depending on the rainfall. Many land users are highly dependent upon national and provincial government to provide funding and resources to ensure sustainable rangeland management and to implement restoration actions. This is especially true for farmers in areas where communal and lease tenure systems are applied, as these farmers are very poor and do not have the funds to implement or maintain sustainable rangeland management systems.

Step 6 constitutes the conclusion of the IAPro process where science and local knowledge, biophysical assessment data and SH perspectives are combined and converged into a collective and integrated evaluation of locally applied restoration and management actions implemented to combat rangeland degradation (Bautista & Orr, 2011). It is important to remember that no absolute best restoration or rangeland management action exists, as the evaluation of the actions depends on tradeoffs between criteria, SH perspectives and farming objectives, the latter varying between commercial and communal farmers, as well as dynamic socio-economic contexts and must be adapted from case to case (Reed *et al*., 2006; Von Maltitz, 2009; Bautista & Orr, 2011; Dreber *et al*., 2013).

4.5 Conclusions

This study showed that the IAPro is a promising tool for a locally-contextualised assessment and evaluation process of restoration and management actions that can be applied to combat rangeland degradation. According to Rojo *et al*. (2012), the PRACTICE approach can play a role in the participatory evaluation of environmental plans and policies to combat rangeland degradation, especially if the structure of the assessment tool is adapted according to a specific study area and the SHs' involvement is positive. The general strength of the IAPro towards participatory evaluation is the bottom-up approach for the assessment of both environmental sustainability and social acceptance and integration. As required by, for instance, the UNCCD, SHs have to be involved in an approach aimed at arriving at feasible solutions and actions in their management strategies – a requirement the IAPro definitely meets (Rojo *et al*., 2012).

Lessons learned in the IAPro and in the specific context of the Molopo study area

Shortcomings and challenges of the IAPro identified by this study for the Molopo farming community include: (a) The fact that the SHs of the Molopo region are already "over-workshopped" due to the many rangeland management projects being conducted in the region, each requiring discussions, meetings and bio-physical and socio-economic inputs and assessments. SHs are continually asked to participate, and consultation fatigue may begin to set in, especially if the SHs perceive their involvement in the project as offering little personal reward and being of no consequence as far as the outcome is concerned (Dreber *et al*., 2013). (b) Due to the huge expanse of the study area, land users have to travel long distances in order to participate in the workshops, which could result in low participation during the participatory steps of the IAPro. (c) It was often very difficult to find sufficient replicate sites for a particular restoration or management action in order to prevent/reduce variability in the data, especially as the IAPro requires that quantitative assessments should only be conducted in the rangelands of the SHs that have been identified and have participated in step 1 of this protocol.

Another aspect that will benefit the study/approach is to ensure the SHs do not only have knowledge regarding the different restoration and management actions but also

implement the actions they have mentioned during the interviews in step 2. (d) It would have been more ideal if both iterations (step 3a and 3b) during the group discussions could have been implemented in a single meeting (Bautista & Orr, 2011); however, due to logistic constraints (distances to travel between farms for quantitative sampling of vegetation), a year passed before the second iteration of indicator ranking could be completed. Consequently, other factors may have caused SHs to change their minds. (e) It is advised to also quantify other indicators (e.g. soil nutrient status) since mostly only productive parameters of the vegetation were assessed during this study. (f) The environmental awareness of several SHs may be influenced by security issues regarding land tenure (Dreber *et al*., 2013). For instance, in a communal rangeland management setup being shared and managed by all, the land user often displays less interest in making an investment in the improvement and protection of natural resources against degradation. In contrast, where a land user that is the owner of the land, he/she is more likely to consider investing money and labour because of the benefits that will be gained as a result of rangeland production (Van Rooyen, 1998) and the future benefits the land will offer his/her progeny likely to inherit the land. In this regard, it should be kept in mind that production improvement may only accrue after many years of restoration (Stocking & Murnaghan, 2001).

The way forward

In South Africa, environmentally and socio-economically accepted restoration and management actions are of importance to combat rangeland degradation, and according to Dreber *et al.* (2013), a modified PRACTICE approach could be a promising tool for the exploration of alternative restoration and management actions not identified by the local members of the multi-stakeholder platform. The work by Reed *et al.* (2007, 2008) provides a monitoring tool for rangeland degradation assessments which can be combined with the PRACTICE approach to improve the evaluation of the best restoration and management actions in a certain region. By combining the two frameworks/approaches, Dreber *et al.* (2013) argue, it might be possible to create a more holistic framework in order to improve decision making by land users and to ensure a more sustainable rangeland management system is embedded in conjunction with a long-term monitoring programme.

Replications and comparisons of the PRACTICE approach are essential to evaluate its suitability and applicability as a participatory approach. The final step of the IAPro (step 7) focuses on participatory activities and will be conducted by distributing what was learned in this study amongst the different local SHs from the Molopo study area during the evaluation process. This knowledge will be shared with the larger global community that is affected by desertification and rangeland degradation through Internet-supported dissemination of audio-visual information (Bautista & Orr, 2011; Rojo *et al*., 2012). See chapter 6, section 6.2.1, for recommendations.

5

The effect of chemical bush control actions and rotational grazing management on the Molopo bushveld vegetation

5.1 Introduction

Please see section 2.3.2 of this thesis (literature review) for an introduction to the issue of bush thickening in southern African savannas. The issue regarding bush thickening that may be affecting the production potential of the Molopo rangelands resulted in the research objectives being formulated, namely (a) how the woody density affects the frequency distribution of grass and woody species and grass functional groups (GFGs), (b) the effect chemical control vs. no control had on the productivity parameters of the vegetation, and (c) whether there is a relationship between the status of the grass layer and woody layer.

Chemical control with arboricides is commonly applied in the Molopo rangelands to mitigate bush thickening. Many new combinations of arboricides have been studied by various people in an attempt to reduce the associated costs and increase the effectiveness of the arboricides on a broad spectrum of woody species (Bovey & McCarty, 1965; Bovey *et al*., 1967). The arboricide available and used by most land users in the Molopo region of South Africa is a systemic arboricide since it is applied to the soil close to the base of the woody plant so that it can be taken up by the roots of the plant. This specific arboricide contains active ingredients such as Tebuthiuron, Ethidimuron or Bromacil (Smit *et al*., 1999). It is marketed as granules, wettable powders or as a liquid, and the concentration of the active ingredient ranges from 20% to 70% (Smit *et al*., 1999, Dube *et al*., 2011). The particular arboricide used by most land users in the Molopo region is called Molopo CC and can either be applied by hand or aeroplane in a granular form (Du Toit & Sekwadi, 2012). The granular form is preferred since it allows for better measuring of dosages (the amount of

granules) and selecting of woody species that have to be controlled. The granules come in grey to bright red colours, making them more visible while lying on the soil surface, thereby preventing the over-use of arboricides (Smit *et al*., 1999).

The advantages of chemical hand control (HC) are, therefore, that less arboricide is used and that it is a more selective method. However, more time is required while it also necessitates the use of workers who are able to distinguish between the different woody plant species and the dosage that has to be applied, especially over large areas. The advantages of aeroplane control (AC) compared to HC are a more uniform application, the ability to treat large areas in less time and the need for fewer labourers (Smit *et al*., 1999). All the woody species, including seedlings, are treated. Often, though, palatable and valuable woody plants are also killed, which may be a disadvantage of this method (Smit *et al*., 1999). Another disadvantage of AC is that it is an expensive process and is often only applied in areas where woody plants have formed very dense stands and where the level of post-treatment production will justify the costs (Smit *et al*., 1999; Dube *et al*., 2011).

5.2 Material and methods

5.2.1 Biophysical assessment

Sampling sites on the rangelands were predefined in step 4 of the IAPro (see section 4.2.1). Only one assessment was conducted per sampling site. A total of 37 quantitative assessments of the woody and grass layer were carried out in January to March 2012, November 2012 (only woody layer) and February to March 2013 (only grass layer) on rangelands varying in management regime (Table 5.1). As it was not possible to determine the initial condition of the vegetation before treatment to combat bush thickening, nearby rangelands under long-term rotational grazing management (RGM) (n = 8) as well as bush-thickened (BT) sites (n = 15) were selected as benchmarks along a gradient from semi-natural savanna in good condition to degraded savanna. For details regarding applied assessment techniques, sampling design and collected vegetation data (i.e. grass forage production, woody phytomass and grass and woody species' relative frequency), please refer to section 4.2.2.

Soil samples were collected from the 30 cm top-soil layer at all sampling sites by making use of a manual soil auger. Litter from the soil surface was removed before soil samples were taken. Nine random soil samples were collected along the two 160 m transects at each sampling site. The soil samples were analysed by the Department of Soil Sciences at the North-West University (NWU), Potchefstroom[7]. Soil chemical analyses included percentage organic carbon, pH (H_2O), pH (KCl) and electrical conductivity (EC). This study formed part of the PRACTICE project and therefore only the latter soil analyses parameters were analysed.

Additional information recorded at each site included site identification name, date of survey and GPS reading, together with photos documenting the vegetation and habitat of the sampling site. A questionnaire was used to gain information from the land user on the type of animals on the farm, stocking rate, the total time period camps were rested, type of chemical treatment (HC or AC), time between treatments and dosage used (Data sheet A4).

Table 5.1: Different chemical control and management actions in the study area: For an explanation of land tenure systems please refer to Table A2.

	Rotational grazing management (RGM)	**Hand chemical control (HC)**	**Aeroplane chemical control (AC)**	**Bush thickened (BT)**
Land tenure type	Commercial and lease farmers	Commercial farmers	Commercial farmers	Commercial and lease farmers
Animals	Cattle	Cattle	Cattle and game	Cattle and game
Treatment	Camps are grazed for 2 weeks and rested for 6 weeks	Selective chemical treatment with Molopo CC granules Reapplication of chemicals every 5 years	Non-selective chemical treatment with Molopo CC granules (2.5 – 3 kg ha^{-1}) Reapplication of chemicals every 10 years	No treatment/ restoration
Stocking density	8-10 ha LSU^{-1}	10 ha LSU^{-1}	10 ha LSU^{-1}	12-20 ha LSU^{-1}

[7] Vermeulen, T., Department of Soil Sciences (Eco-Analytica, NWU), PO Box 19140, Noordbrug, Potchefstroom 2522. Tel: 018 293 3900.

5.2.2 Calculations

A leaf quantification technique (BECVOL-model by Smit (1989[b])) was used to determine the total dry matter yield of the woody layer based on the biometric measures (see section 4.2.2.2). The browsing capacity was calculated for both the < 2 m and > 2 m woody height class, as smaller browser animal species such as goats and impala are only able to utilise the available browse < 2 m, whereas for giraffes, the browse-able material in the > 2 m height class is relevant (Trollope, 2011). The browsing capacity was calculated using the formula $d \times (DM \times f \times r^{-1})^{-1}$ and was expressed in ha Browser Unit^{-1} (BU) (Smit, 2006). In this formula, *d* is the total number of days in a year (365), *DM* the total leaf dry matter yield ha^{-1} of palatable woody plants, *f* the utilisation factor (the average of 0.35 (35%) is commonly used) and *r* the daily leaf dry matter required per browser unit (2.5% of body mass for a Kudu antelope with an average weight of 140 kg, thus 3.5 kg day^{-1}). A value regarding the phenology can be added to the formula above as an additional variable. However, in vegetation types that are dominated by a woody layer of heterogeneous deciduous species a more accurate approach would be to apply the browsing capacity formula as shown above without the incorporation of values assigned to phenology (See Smit, 2006).

The calculation of grazing capacity (ha LSU^{-1}), grass forage production (kg ha^{-1}) and woody plant phytomass (tree equivalents (TE) ha^{-1}) followed, using Teague *et al*. (1981) as described in section 4.2.2.

5.2.2.1 Statistical analysis

Before any formal statistical analysis was carried out, preliminary checks of the data were done (i.e. box plots were examined with respect to symmetry of distributions and outliers, and data was log-transformed if necessary) (Quinn & Keough, 2002). If the assumption of variance homogeneity has still not been met, a non-parametric test (i.e. the Kruskal-Wallis test) had to be used (Quinn & Keough, 2002). Comparisons of effects of the restoration (i.e. HC vs. AC) and management (i.e. RGM) actions on different productivity parameters of the vegetation (e.g. grass forage production as well as grazing and browsing capacity) and soil parameters were done using one-way analysis of variance (ANOVA) with post-hoc Tuckey's honestly significant difference (HSD) test (Quinn & Keough, 2002) for pair-wise

comparisons. All differences were considered significant at $p < 0.05$. Relationships between grass forage production, grazing capacity and woody plant densities were established using the data from all the assessment sites ($n = 37$) in a simple linear regression analysis (Hammer *et al*., 2001). Regression analyses were conducted using STATISTICA 11 (STATSOFT, Inc., 2009).

The Kruskal-Wallis test (Kruskal & Wallis, 1952) with Mann-Withney post-hoc pair-wise test (Zar, 1996) was carried out to compare and determine statistical differences in the frequency distributions of grass functional groups (GFGs) and grass and woody species between the various actions. See Tables A3 and A4 for the grass species allocations to the six GFG types which were grouped according to the plant's life cycle, palatability and response type as described in Van Oudtshoorn's (2012) guide to the classification of grass species for southern Africa, personal observations and as noted by local land users regarding the palatability and plant vigour of a particular species. All differences were considered significant at $p < 0.05$.

The woody and grass vegetation component and abiotic variables were assessed to analyse similarities in overall species composition and underlying environmental gradients by way of a multivariate ordination technique. First, a de-trended correspondence analysis (DCA) was conducted using CANOCO (Ter Braak, 1992) in order to check the length of gradient and to decide which model (i.e. linear or unimodal) would be best suited. It was decided to perform an indirect Principal Component Analysis (PCA) to identify patterns in species composition and to confirm pre-defined actions. Ordination axes could be extracted to describe the correlations between the species composition, abiotic variables and the management and restoration actions. Abiotic variables (i.e. soil organic carbon, EC, pH and woody phytomass) and nominal variables (i.e. game farms and cattle farms as well as restoration and management actions) were used as supplementary data. All the grass and woody species were used in the ordination.

Significant differences in the plant species composition between restoration and management actions were identified by calculating dissimilarities between the actions (i.e. RGM, HC, AC and BT – Table 5.1) using the analysis of similarity (ANOSIM, see Clarke, 1993). ANOSIM considers the composition and species frequencies of grass and woody plants and is a non-parametric test of significant

differences between two actions, based on any distance measure. Distances are converted to ranks (Hammer *et al*., 2001). Rank dissimilarities were used based on the Bray-Curtis coefficient of similarity and indicated how separated distinct predefined actions were. It also calculated an R-statistic on a scale of 0 (actions are indifferent) to 1 (i.e. actions are totally different). A similarity percentage analysis (SIMPER, see Clarke, 1993) was performed in order to identify the species that account most for the dissimilarities between the actions given by ANOSIM. All analyses were conducted using the software PAST v.2.15 (Hammer *et al*., 2001).

5.3 Results

5.3.1 General patterns in plant species composition

The indirect PCA ordination was carried out to identify environmental gradients in species distribution patterns and to explain the variation in vegetation composition among the different restoration and management actions and bush-thickened (BT) sites. The eigenvalue of the first ordination axis of (0.472) indicated that differences in the vegetation composition, management type and woody phytomass between the various plots of the restoration and management actions were strongly correlated to the first axis. The cumulative variance explained by the second ordination axis for the species-environment relation was 75.2% and 60.7% for the cumulative species data variance.

The PCA revealed a reasonable degradation gradient on the first ordination axis from the grass species *Aristida stipitata* and *Schmidtia kalahariensis,* with very low potential grazing values, together with high abundances of the encroacher species *Acacia mellifera* in the BT sites, which were also characterised by a high woody phytomass (Figure 5.1). The perennial, mostly palatable, climax grass species *Schmidtia pappophoroides* and the more palatable woody plant species *Acacia erioloba* were the most abundant in rotational grazing management (RGM) and hand control (HC) sites, which were characterised by a lower woody phytomass (Figure 5.1). RGM and HC sites formed clusters and appeared the most similar in species composition. The aeroplane control (AC) cluster was mostly characterised by the woody plant species *Boscia albitrunca* and the less palatable, sub-climax grass species *Eragrostis lehmanniana* and *Melinis repens* (Figure 5.1). These separate

cluster formations of the different actions illustrated in Figure 5.1 confirm the groupings of the pre-defined restoration (HC and AC) and management (RGM) actions.

Soil variables only played a minor role in the species distribution patterns among the actions as indicated by the short arrows (Figure 5.1). This is in line with the finding of no significant differences in all assessed soil parameters (Figure A13, in Appendix).

According to the results, game farms were found to be mostly associated with the woody and less palatable grass species most commonly found in the BT sites, with more browse-able material. Cattle farms, on the other hand, were mostly associated with the more palatable grass and woody species associated with RGM and HC sites (Figure 5.1).

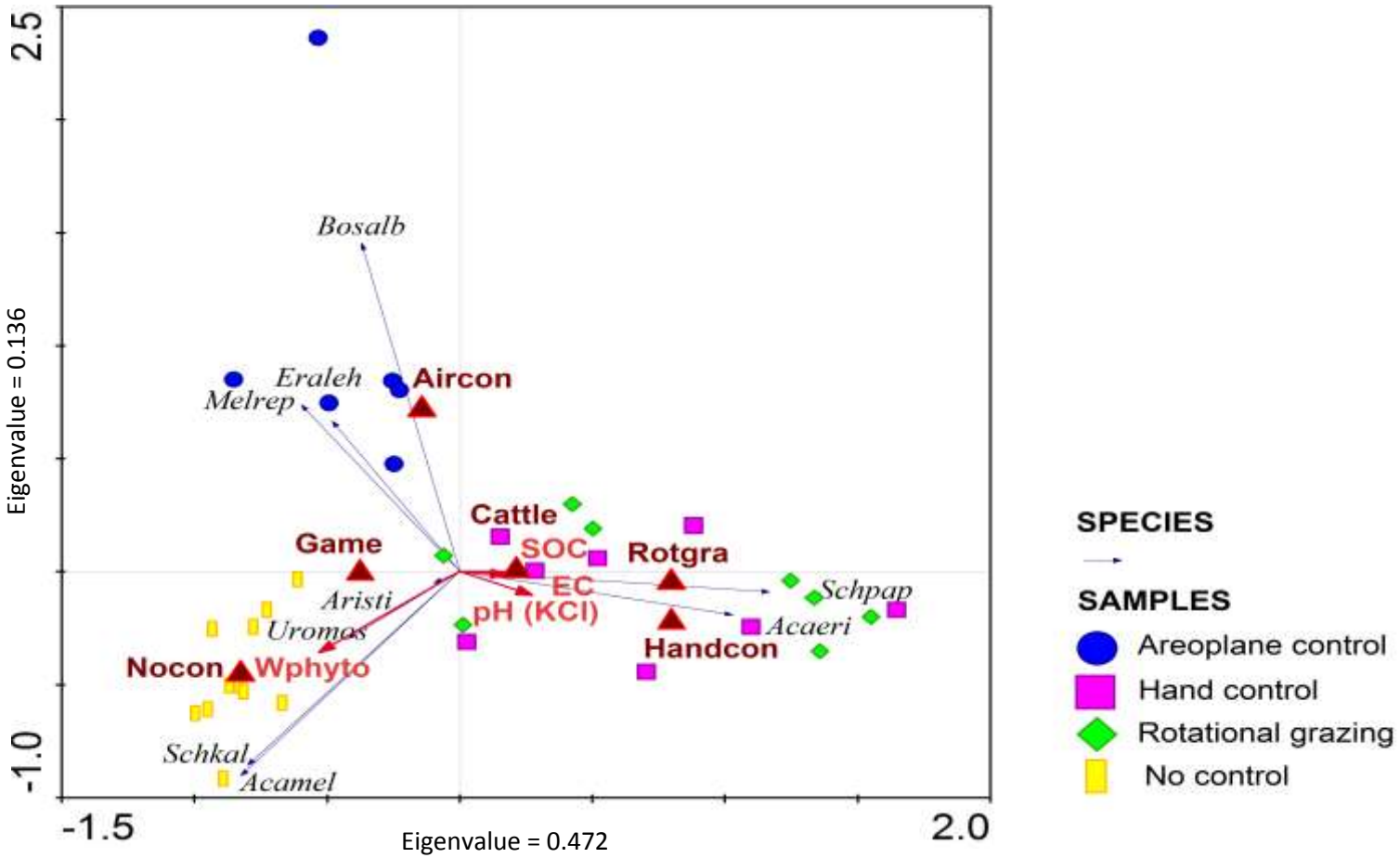

Figure 5.1: PCA ordination tri-plot indicating the correlation between the grass and woody species composition of the sampling plots in terms of environmental variables and management type: ***Acaeri*** – *Acacia erioloba,* ***Acamel*** – *Acacia mellifera,* ***Bosalb*** – *Boscia albitrunca,* ***Eraleh*** – *Eragrostis lehmanniana,* ***Melrep*** – *Melinis repens,* ***Schkal*** – *Schmidtia kalahariensis,* ***Schpap*** – *S. pappophoroides,* ***Uromos*** – *Urochloa mosambicensis*. Red arrows indicate supplementary environmental variables. The brown triangles are nominal environmental variables. **Aircon** – Aeroplane control, **EC** – Electrical conductivity, **Handcon** – Hand control, **Nocon** – Bush-thickened sites, **Rotgra** – Rotational grazing, **SOC** – Soil organic carbon.

All the grass and woody species were used in the ordination, but for the sake of clarity, only the most abundant species are indicated in the graph. Environmental variables included soil organic carbon, electrical conductivity and $pH_{(KCl)}$.

5.3.2 Frequency distribution of grass species

From the results regarding the frequency distribution of grass species, as indicated in the PCA, a significant higher occurrence of perennial, palatable, climax type species ($p < 0.001$), notably *Schmidtia pappophoroides,* was found in the HC and RGM sites (Table 5.2) compared to BT sites. Less palatable, sub-climax grasses, such as *Melinis repens* (9.8 ± 6.3%) and *Eragrostis lehmanniana* (27.6 ± 19.0%), were found to be significantly higher ($p < 0.001$) in the AC sites compared to RGM and HC sites respectively. Although not significant, *Schmidtia pappophoroides* had a higher abundance in AC sites (20.7 ± 22.3%), compared to the 1.5 ± 3.6% found in BT sites (Table 5.2). There was a significant higher occurrence of annual, unpalatable, pioneer species ($p < 0.001$) such as *S. kalahariensis* (22.6 ± 15.1%) in the BT sites (Table 5.2).

Table 5.2: Effect of RGM (n = 8), chemical HC (n = 7) and chemical AC (n = 7) actions compared to BT sites (n = 15) on the relative frequency distribution (%) of grass species: Means (±SD) with different lower-case letters in a row indicate a significant difference at $p < 0.001$ (Kruskal-Wallis test with Mann-Whitney post-hoc pair-wise test).

	Rotational grazing management (RGM)	Hand chemical control (HC)	Aeroplane chemical control (AC)	Bush thickened (BT)
Anthephora argentea	0.9±2.1	0.0±0.0	0.0±0.0	0.0±0.0
Anthephora pubescens	1.8±5.0	0.0±0.0	0.0±0.0	0.0±0.0
Aristida congesta	0.1±0.4^{a}	0.1±0.4^{a}	0.9±2.3 ^{a}	0.9±2.3^{a}
Aristida meridionalis	0.1±0.4^{a}	0.1±0.4^{a}	0.3±0.8 ^{a}	1.0±2.6^{a}
Aristida stipitata	12.0±6.5^{a}	4.3±7.2^{a}	3.3±3.2^{a}	20.7±20.6^{a}
Brachiaria nigropedata	1.1±3.2^{a}	0.6±1.5^{a}	1.7±4.5^{a}	0.0±0.0
Centropodia glauca	1.4±1.9^{a}	0.3±0.8^{a}	0.0±0.0	0.0±0.0
Digitaria eriantha	0.0±0.0	0.0±0.0	0.1±0.4	0.0±0.0
Eragrostis lehmanniana	14.5±7.4^{a,b}	6.4±6.7^{a}	27.6±19.0^{b}	19.3±15.3^{a,b}
Eragrostis nindensis	0.0±0.0	0.0±0.0	0.0±0.0	0.1±0.3^{a}
Eragrostis pallens	4.6±8.6^{a}	0.0±0.0	0.0±0.0	0.7±2.1^{a}
Eragrostis trichophora	0.4±1.1^{a}	0.4±1.1^{a}	2.1±2.3^{a}	2.2±3.5^{a}
Megaloprotachne albescens	0.0±0.0	0.0±0.0	0.4±1.1	0.0±0.0
Melinis repens	0.3±0.7^{a}	3.2±3.9^{a,b}	9.8±6.3^{b}	5.3±5.5^{a,b}
Pogonarthria squarrosa	0.1±0.4^{a}	0.0±0.0	0.9±1.2^{a}	0.1±0.6^{a}
Schmidtia pappophoroides	56.7±23.0^{a}	59.1±25^{a}	20.7±22.3^{a,b}	1.5±3.6^{b}
Schmidtia kalahariensis	0.0±0.0	0.7±1.3^{a}	0.1±0.4^{a}	22.6±15.1^{b}
Stipagrostis hirtigluma	0.6±1.8	0.0±0.0	0.0±0.0	0.0±0.0
Stipagrostis uniplumis	4.9±7.9^{a}	9.8±7.6^{a}	17.9±10.6^{a}	12.7±9.4^{a}
Tragus berteronianus	0.3±0.7^{a}	0.0±0.0	0.3±0.8^{a}	0.3±1.1^{a}
Triraphis schinzii	0.1±0.4^{a}	0.0±0.0	0.0±0.0	1.4±3.5^{a}
Urochloa mosambicensis	0.1±0.4^{a}	14.9±15.6^{a}	13.7±15.0^{a}	11.1±17.2^{a}

5.3.3 Frequency distribution of woody species

There was a significantly higher occurrence ($p < 0.001$) of the palatable woody plant species *Acacia erioloba* in the RGM (40.8 ± 33.1%; relative frequency values) and HC sites (28.1 ± 18.6%) than in the AC and BT sites (Table 5.3). The dominant species in BT sites was the species *Acacia mellifera* (38.2 ± 22.9%) followed by *Grewia flava* (21.7 ± 11.1%) (Table 5.3). In the selective HC sites, very low numbers of the species *A. mellifera* (17 ± 14.1%) and *Dichrostachys cinerea* (0.8 ± 1.4%) occurred. The dominant species resulting from the non-selective application of chemicals by AC was *Boscia albitrunca* (37.7 ± 26.6%) (Table 5.3).

Table 5.3: Effect of RGM (n = 8), chemical HC (n = 7) and chemical AC (n = 7) actions compared to BT sites (n = 15) on the woody plant species composition on relative frequency (%) (only showing woody plant species with a relative frequency above 5%; for a detailed list, see Table A5): Means (±SD) with different lower-case letters in a row indicate a significant difference at $p < 0.001$ (Kruskal-Wallis test with Mann-Whitney post-hoc pair-wise test).

	Rotational grazing management (RGM)	Hand chemical control (HC)	Aeroplane chemical control (AC)	Bush thickened(BT)
Acacia erioloba	40.8±33.1[a]	28.1±18.6[a]	10.1±13.5[a,b]	2.5±3.2[b]
Acacia luederitzii	5.0±5.6[a]	0.0±0.0	5.3±5.2[a]	5.9±7.2[a]
Acacia mellifera	12.7±13.8[a]	17.0±14.1[a]	15.1±17.0[a]	38.2±22.9[a]
Boscia albitrunca	12.7±13.8[a]	3.6±4.8[a]	37.7±26.6[b]	3.8±3.2[a]
Dichrostachys cinerea	10.5±18.8[a]	0.8±1.4[a]	1.9±2.7[a]	10.8±19.6[a]
Grewia flava	16.8±20.8[a]	33.6±17.8[a]	17.3±17.3[a]	21.7±11.1[a]
Terminalia sericea	0.7±1.4[a]	0.0±0.0	0.0±0.0	5.3±10.9[a]

5.3.4 Similarities between restoration and management actions and bush-thickened sites regarding the vegetation composition

After the confirmation of clear groupings in the predefined actions, ANOSIM was applied to establish the degree of variance between the respective actions. Results from the ANOSIM confirmed the results from the PCA, namely that there was a significantly high dissimilarity in plant species composition between RGM and BT sites as indicated by the *post hoc R*-statistics on the species assemblages of group comparisons as described in Table 5.4. A significantly high dissimilarity in plant species composition was found between RGM and AC sites (68%), as well as between HC and BT sites (65.3%) (Table 5.4). Species assemblages of RGM and

HC were barely separable with the lowest mean dissimilarities (51.5%) calculated by SIMPER for the respective action comparisons (Tables 5.4 and 5.8).

Table 5.4: ANOSIM results for direct comparison between different restoration and management actions and BT sites and the overall mean dissimilarity as calculated by SIMPER: RGM = Rotational grazing management, BT = Bush thickened, HC = Hand control, AC = Aeroplane control. The *R*-statistic indicated on a scale whether actions vary from one another or not (from 0 – actions are indifferent to 1 – actions are totally different).

Treatments	*R*	*p*	Mean dissimilarity (%)
RGM vs. BT	0.9	0.0006	74.5
HC vs. BT	0.9	0.0006	65.3
AC vs. BT	0.7	0.0012	61.1
RGM vs. HC	0.1	0.5424	51.5
RGM vs. AC	0.5	0.0102	68.0
HC vs. AC	0.4	0.0360	60.2

As calculated by SIMPER, only 12 species were responsible for contributing 87% to the total dissimilarity between AC and BT sites. The woody shrub, *Acacia mellifera,* and the woody tree, *Boscia albitrunca,* accounted for most of the dissimilarity (Table 5.5). A discriminating species for BT sites was the annual, mostly unpalatable, pioneer grass species *Schmidtia kalahariensis*, whose mean percentage contribution to the overall dissimilarity was the highest for grasses (Table 5.5). Another woody species, *A. luederitzii,* and the mostly unpalatable, pioneer sub-climax grass species *Aristida stipitata* found in BT sites also resorted under the top 12 species; its mean percentage contribution to the overall dissimilarity was, however, low (Table 5.5).

Table 5.5: Top 12 plant species cumulatively accounting for 87% of vegetation dissimilarity between AC and BT sites as calculated by SIMPER: 'Contribution %' indicates the average percentage contribution from the ith species to the overall Bray-Curtis-dissimilarity between BT and AC sites. "Cumulative %" indicates the cumulative percentage contribution to the total dissimilarity (tabled here as a cut-off of 87% for the total of 41 identified species; see Table A6).

Species	Mean relative frequency		Contribution (%)	Cumulative (%)
	Bush thickened (BT)	Aeroplane control (AC)		
Acacia mellifera	47.1	15.1	13.6	13.6
Boscia albitrunca	5.2	37.7	13.4	27.0
Schmidtia kalahariensis	28.9	0.1	11.8	38.8
Schmidtia pappophoroides	1.5	20.7	8.3	47.1
Eragrostis lehmanniana	18.1	27.6	7.7	54.8
Grewia flava	23.0	17.3	7.0	61.9
Urochloa mosambicensis	14.9	13.7	6.9	68.8
Stipagrostis uniplumis	16.6	17.9	4.6	73.4
Aristida stipitata	10.7	3.3	4.1	77.5
Acacia erioloba	3.6	10.1	3.8	81.2
Melinis repens	6.3	9.8	3.0	84.2
Acacia luederitzii	7.6	5.3	2.7	86.7

Only 13 species contributed to 87% of total dissimilarity between RGM and BT sites (Table 5.6). The perennial, mostly palatable, climax grass species *Schmidtia pappophoroides* and the palatable woody plant species *Acacia erioloba* accounted for most of the dissimilarity (Table 5.6). The woody species *A. mellifera* was the third species that accounted most for the dissimilarity with a high occurrence in the BT sites (Table 5.6). The perennial, less palatable, pioneer to sub-climax grass *Melinis repens* and woody plant *Boscia albitrunca* were also under the top 13 species; their mean percentage contribution to the overall dissimilarity was, however, low (Table 5.6).

Table 5.6: Top 13 plant species cumulatively accounting for 87% of vegetation dissimilarity between RGM and BT sites as calculated by SIMPER: 'Contribution %' indicates the average percentage contribution from the ith species to the overall Bray-Curtis-dissimilarity between BT and RGM sites. "Cumulative %" indicates the cumulative percentage contribution to the total dissimilarity (tabled here as a cut-off of 87% for the total of 41 identified species; see Table A7).

Species	**Mean relative frequency**		**Contribution (%)**	**Cumulative (%)**
	Bush thickened (BT)	Rotational grazing management (RGM)		
Schmidtia pappophoroides	0.5	56.7	18.9	18.9
Acacia erioloba	3.6	40.8	12.8	31.6
Acacia mellifera	47.1	12.7	11.8	43.4
Schmidtia kalahariensis	28.9	0.0	9.7	53.1
Grewia flava	23.0	16.8	6.6	59.8
Urochloa mosambicensis	14.9	0.3	5.0	64.8
Stipagrostis uniplumis	16.6	4.9	4.7	69.5
Eragrostis lehmanniana	18.1	14.5	4.1	73.5
Aristida stipitata	10.7	11.9	3.7	77.2
Dichrostachys cinerea	1.5	10.4	3.5	80.7
Acacia luederitzii	7.6	5.0	2.3	83.0
Melinis repens	6.3	0.3	2.1	85.1
Boscia albitrunca	5.2	0.8	1.6	86.7

Table 5.7: Top 11 plant species cumulatively accounting for 88% of vegetation dissimilarity between BT and HC sites as calculated by SIMPER: 'Contribution %' indicates the average percentage contribution from the ith species to the overall Bray-Curtis-dissimilarity between BT and HC sites. "Cumulative %" indicates the cumulative percentage contribution to the total dissimilarity (tabled here as a cut-off of 88% for the total of 41 identified species; see Table A8).

Species	Mean relative frequency		Contribution (%)	Cumulative (%)
	Bush thickened (BT)	Hand control (HC)		
Schmidtia pappophoroides	0.5	59.1	22.5	22.5
Acacia mellifera	47.1	17.0	11.9	34.3
Schmidtia kalahariensis	28.9	0.7	10.8	45.2
Acacia erioloba	3.6	28.1	9.5	54.7
Grewia flava	23.0	33.6	6.6	61.3
Urochloa mosambicensis	14.9	14.9	6.1	67.3
Eragrostis lehmanniana	18.1	6.4	5.4	72.8
Stipagrostis uniplumis	16.6	9.8	4.3	77.1
Aristida stipitata	10.7	4.4	4.1	81.2
Rhigozum brevispinosum	2.7	7.7	3.5	84.6
Acacia luederitzii	7.6	0.0	2.9	87.5

All comparisons of the restoration and management actions with BT sites revealed that, typically, encroacher woody species, such as *Acacia mellifera;* the unpalatable, annual grass specie, *Schmidtia kalahariensis* and the more valued woody, often palatable, woody and grass species *A. erioloba* and *S. pappophoroides* accounted most for the dissimilarity in species assemblages (Tables 5.5 to 5.7). Similar results are reflected in the PCA ordination (Figure 5.1). Palatable climax grasses, such as *Schmidtia pappophoroides*, were generally more abundant in RGM and HC sites, characterising a better rangeland condition (Van Oudtshoorn, 2012) (Table 5.7). Similar to the clusters in the PCA ordination, RGM and HC were always associated with one another (Figure 5.1 and Table 5.8) reflecting similar plant species compositions. RGM and HC differed mostly from AC in the high occurrence of the perennial, mostly palatable, climax grass species *Schmidtia pappophoroides* in the RGM and HC sites and the dominance of the woody plant species *Boscia albitrunca* in AC sites, which accounted for most of the dissimilarity (Tables 5.9 and 5.10).

Table 5.8: Top 14 plant species cumulatively accounting for 87% of vegetation dissimilarity between RGM and HC sites as calculated by SIMPER: 'Contribution %' indicates the average percentage contribution from the ith species to the overall Bray-Curtis-dissimilarity between RGM and HC sites. "Cumulative %" indicates the cumulative percentage contribution to the total dissimilarity (tabled here as a cut-off of 87% for the total of 41 identified species; see Table A9).

Species	**Mean abundance**		**Contribution (%)**	**Cumulative (%)**
	Hand control (HC)	Rotational grazing management (RGM)		
Acacia erioloba	28.1	40.8	14.9	14.9
Schmidtia pappophoroides	59.1	56.7	13.1	28.0
Grewia flava	33.6	16.8	12.3	40.3
Acacia mellifera	17	12.7	7.3	47.6
Urochloa mosambicensis	14.9	0.1	7.2	54.8
Dichrostachys cinerea	0.8	10.4	5.1	59.9
Eragrostis lehmanniana	6.4	14.5	5.0	64.9
Aristida stipitata	4.4	11.9	4.9	69.8
Stipagrostis uniplumis	9.8	4.9	4.5	74.3
Rhigozum brevispinosum	7.7	0	3.7	78.0
Acacia haematoxylon	1.8	4.4	2.4	80.5
Melinis repens	6.3	0.3	2.1	85.1
Boscia albitrunca	5.2	0.8	1.6	86.7

Table 5.9: Top 11 plant species cumulatively accounting for 87% of vegetation dissimilarity between AC and HC sites as calculated by SIMPER: 'Contribution %' indicates the average percentage contribution from the ith species to the overall Bray-Curtis-dissimilarity between AC and HC sites. "Cumulative %" indicates the cumulative percentage contribution to the total dissimilarity (tabled here as a cut-off of 87% for the total of 41 identified species; see Table A10).

Species	**Mean abundance**		**Contribution (%)**	**Cumulative (%)**
	Aeroplane control (AC)	Hand control (HC)		
Schmidtia pappophoroides	20.7	59.1	17.6	17.6
Boscia albitrunca	37.7	3.6	14.3	31.9
Grewia flava	17.3	33.6	9.6	41.5
Acacia erioloba	10.1	28.1	9.4	50.9
Eragrostis lehmanniana	27.6	6.4	9.1	59.9
Acacia mellifera	15.1	17.0	6.9	66.8
Urochloa mosambicensis	13.7	14.9	6.1	72.9
Stipagrostis uniplumis	17.9	9.8	4.8	77.8
Rhigozum brevispinosum	2.5	7.7	3.6	81.3
Melinis repens	9.8	3.2	3.3	84.7
Aristida stipitata	3.3	4.4	2.2	86.9

Table 5.10: Top 13 plant species cumulatively accounting for 86% of vegetation dissimilarity between RGM and AC sites as calculated by SIMPER: 'Contribution %' indicates the average percentage contribution from the ith species to the overall Bray-Curtis-dissimilarity between RGM and AC sites. "Cumulative %" indicates the cumulative percentage contribution to the total dissimilarity (tabled here as a cut-off of 87% for the total of 41 identified species; see Table A11).

Species	Mean abundance		Contribution (%)	Cumulative (%)
	Aeroplane control (AC)	Rotational grazing management (RGM)		
Schmidtia pappophoroides	15.0	56.7	20.7	15.0
Boscia albitrunca	13.6	0.8	37.7	28.6
Acacia erioloba	13.1	40.8	10.1	41.7
Grewia flava	7.2	16.8	17.3	48.9
Eragrostis lehmanniana	6.0	14.5	27.6	54.9
Acacia mellifera	5.8	12.7	15.1	60.7
Stipagrostis uniplumis	5.6	4.9	17.9	66.3
Urochloa mosambicensis	5.0	0.1	13.7	71.3
Dichrostachys cinerea	3.8	10.4	1.9	75.1
Melinis repens	3.5	0.3	9.8	78.7
Aristida stipitata	3.5	11.9	3.3	82.2
Acacia luederitzii	2.1	5.0	5.3	84.3
Eragrostis pallens	1.7	4.6	0.0	86.0

5.3.5 Frequency distribution of grass functional groups (GFGs)

As mentioned in Table A4, the grass species were divided into six grass functional group (GFG) types. GFG types were similar across the different restoration and management actions and BT sites, as was found in the PCA (Figures 5.1 and 5.2). By and large, the grass layer, 61.9 ± 16.5% and 60.0 ± 27.2% in the RGM and HC sites respectively, was dominated by perennial, more palatable (high grazing value), climax grass type species (GFG 1 including species such as *Schmidtia pappophoroides, Brachiaria nigropedata* and *Anthephora pubescence*) (Table A4). GFG 1's relative frequency distribution in the AC sites was significantly lower (19.1 ± 21.1%), compared to RGM and HC sites ($p < 0.001$). This group (GFG 1) also had the lowest relative frequency distribution (0.5 ± 1.5%) in BT sites ($p < 0.001$ Figure 5.2).

GFG 2, which included species such as *Eragrostis lehmanniana, E. nindensis* and *Stipagrostis uniplumis,* was the dominant GFG in the AC and BT sites and had a significantly higher relative frequency distribution ($p < 0.001$) compared to RGM and HC (Figure 5.2 and Table A12). GFG 5 included species such as *Melinis repens* and *Pogonarthria squarrosa,* and this group was significantly higher in AC sites (10.8 ± 6.9%) compared to the other actions. GFG 6 (including species such as *Schmidtia kalahariensis* and *Tragus berteronianus,* compare Table A4) was significantly higher in the BT sites (28.9 ± 9.9%) compared to the HC, AC and RGM actions. Grass species included in GFGs 5 and 6 are indicative of poor rangeland conditions compared to the grass species in GFG 1, which are highly productive species (Van Oudtshoorn, 2012).

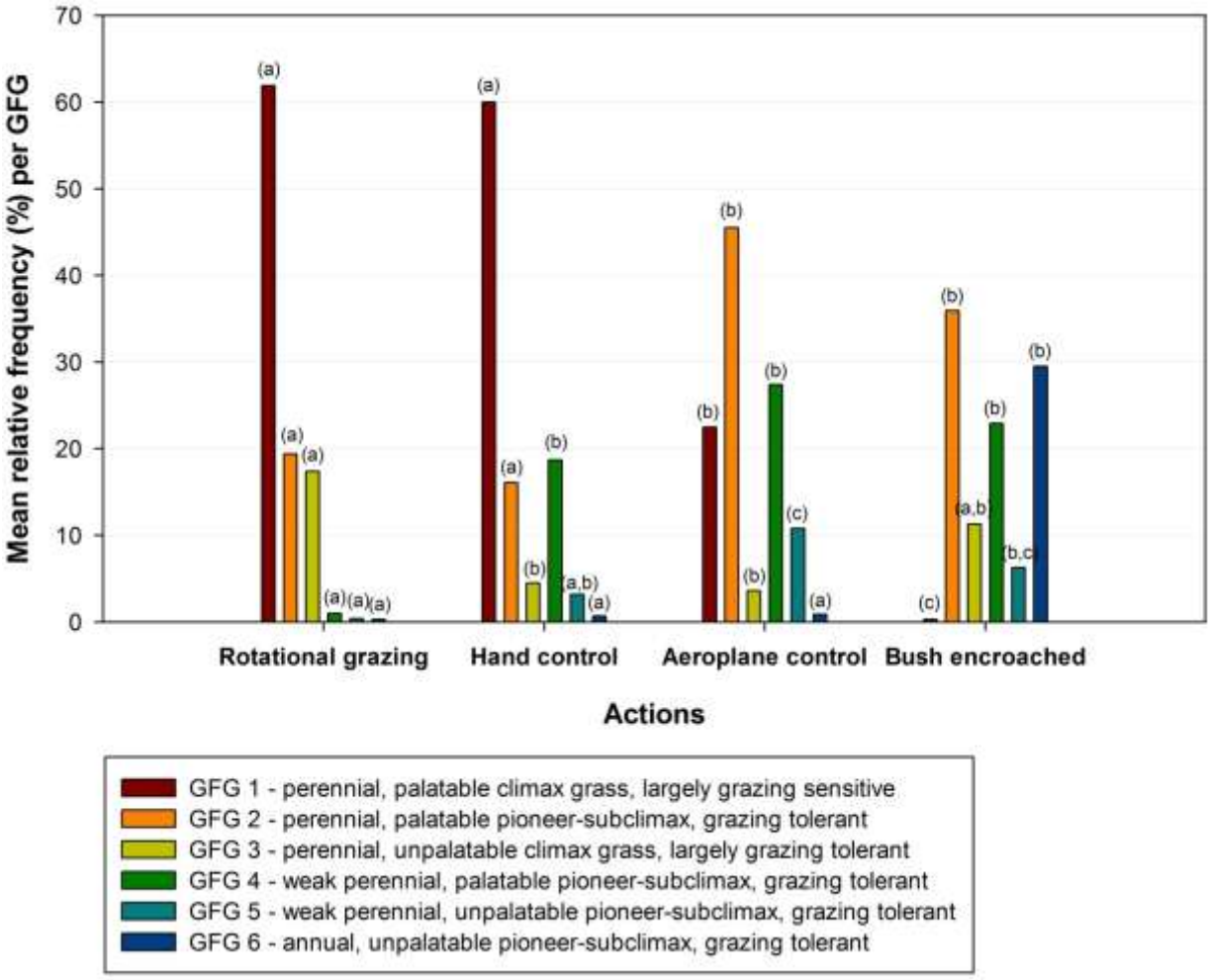

Figure 5.2: Effect of rotational grazing (n = 8), chemical hand control (n = 7) and chemical aeroplane control (n = 7) actions compared to bush-thickened sites (n = 15) on the relative frequency distribution of grass functional groups (GFGs): Different lower-case letters in a row indicate a significant difference at $p < 0.001$ (Kruskal-Wallis test with Mann-Whitney post-hoc pair-wise test).

5.3.6 Effect of restoration and management actions on density and productivity parameters of the grass and woody layers

5.3.6.1 Grass layer

In the RGM sites, a desirable and significantly higher grazing capacity (7.1 ± 1.6 ha LSU^{-1}) ($p < 0.05$) was found compared to the BT sites. The higher grazing capacity related to significantly higher forage production (Figure 5.3a) and the overall higher grazing value of grass species present (compare Table 5.2). Due to this significantly higher grass forage production and grazing capacity, the RGM sites were selected as the "benchmark" sites for good or better rangeland conditions. The second highest average grazing capacity (9.9 ± 3.2 ha LSU^{-1}) was found in the HC sites, which was significantly higher ($p < 0.05$) when compared to the BT sites. This may be due to the abundance of the perennial, palatable, climax grass species *Schmidtia pappophoroides* (compare Table 5.2) and significantly high grass forage production ($p < 0.05$) (Figure 5.3a). In the AC sites, a lower grazing capacity (10.5 ± 2.3 ha LSU^{-1}) was found if compared to RGM and HC sites (Figure 5.3b). Although not significant, this could be due to a high average frequency of less palatable, perennial to weak perennial (pioneer-sub-climax) grass species, such as *Eragrostis lehmanniana* and *Stipagrostis uniplumis* (compare Table 5.2).

The grazing capacity of the BT sites was the lowest with 94.5 ± 51.3 ha LSU^{-1} ($p < 0.05$) (Figure 5.3a). The low grazing capacity resulted from the low to medium grazing value of the majority of grasses found under bush-thickened conditions, which constituted 62.6% of the grass layer (i.e. species *Aristida stipitata, Eragrostis lehmanniana* and *Schmidtia kalahariensis* (compare Table 5.2)).

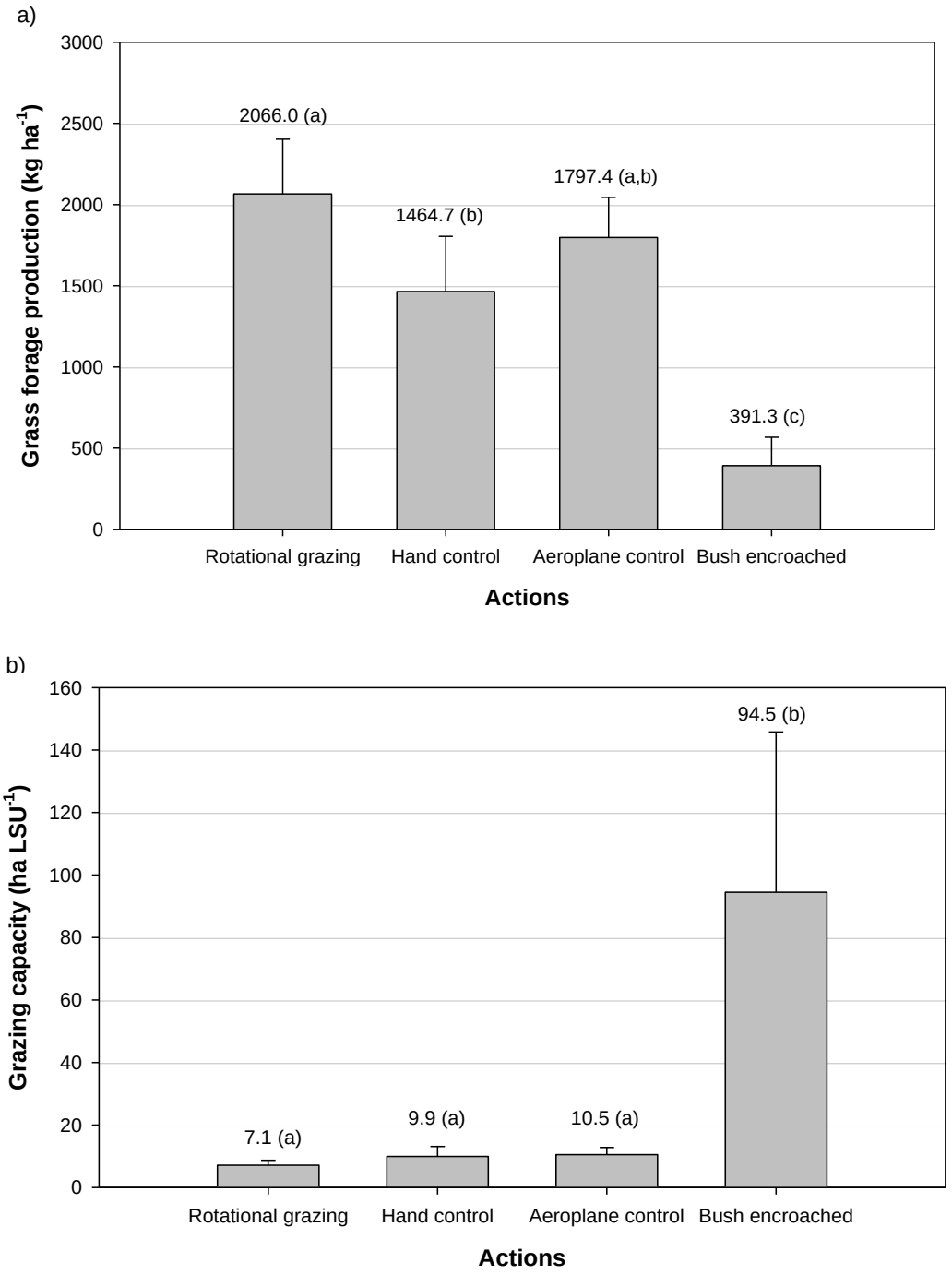

Figure 5.3: Effect of rotational grazing (n = 8), chemical hand control (n = 7) and chemical aeroplane control (n = 7) actions compared to bush-thickened sites (n = 15) on (a) the grass forage production (kg ha^{-1}) and (b) grazing capacity (ha LSU^{-1}): Bars indicate mean values with standard deviations. Different lower-case letters indicate a significant difference at $p < 0.05$ (ANOVA with post-hoc Tukey's HSD test).

5.3.6.2 Woody layer

A significantly higher ($p < 0.05$) woody phytomass was found in the BT sites, compared to the RGM, HC and AC sites (Figure 5.4). Due to this significantly higher woody phytomass, the BT sites were selected as the alternative "benchmark" sites for poor or lower rangeland conditions. The browsing capacity for both height classes (< 2 m and > 2 m) in the BT sites was significantly higher ($p < 0.05$, Figure 5.5) due to the higher availability of browse-able material. AC was found to have a significantly lower woody phytomass (200.3 ± 57.1 TE ha^{-1}) ($p < 0.05$) compared to the BT sites (Figure 5.4). Having the lowest woody phytomass, AC consequently also had the lowest browsing capacity ($p < 0.05$), especially in the < 2 m woody height class (480.9 ± 295.9 ha BA^{-1}) compared to the BT sites (9.7 ± 3.3 ha BA^{-1}) that have not been chemically controlled (Figure 5.5).

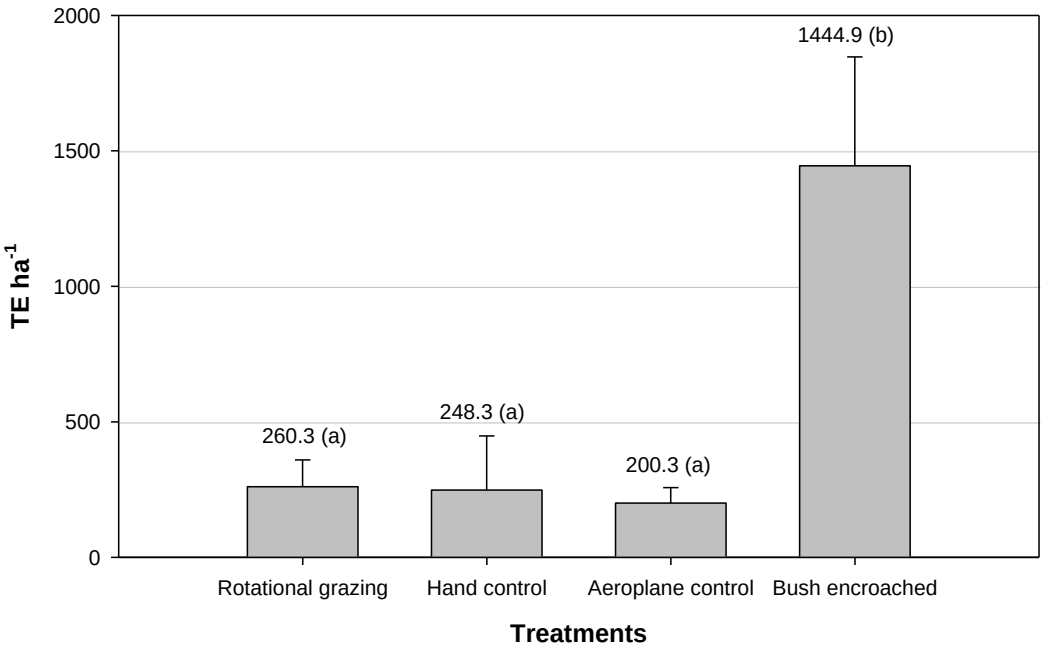

Figure 5.4 Effect of rotational grazing (n = 8), chemical hand control (n = 7) and chemical aeroplane control (n = 7) actions compared to bush-thickened sites (n = 15) on the woody phytomass (tree equivalents ha^{-1} (TE ha^{-1})) for both woody height classes (< 2m and > 2 m): Bars indicate mean values with standard deviations. Different lower-case letters indicate a significant difference at $p < 0.05$ (ANOVA with post-hoc Tukey's HSD test).

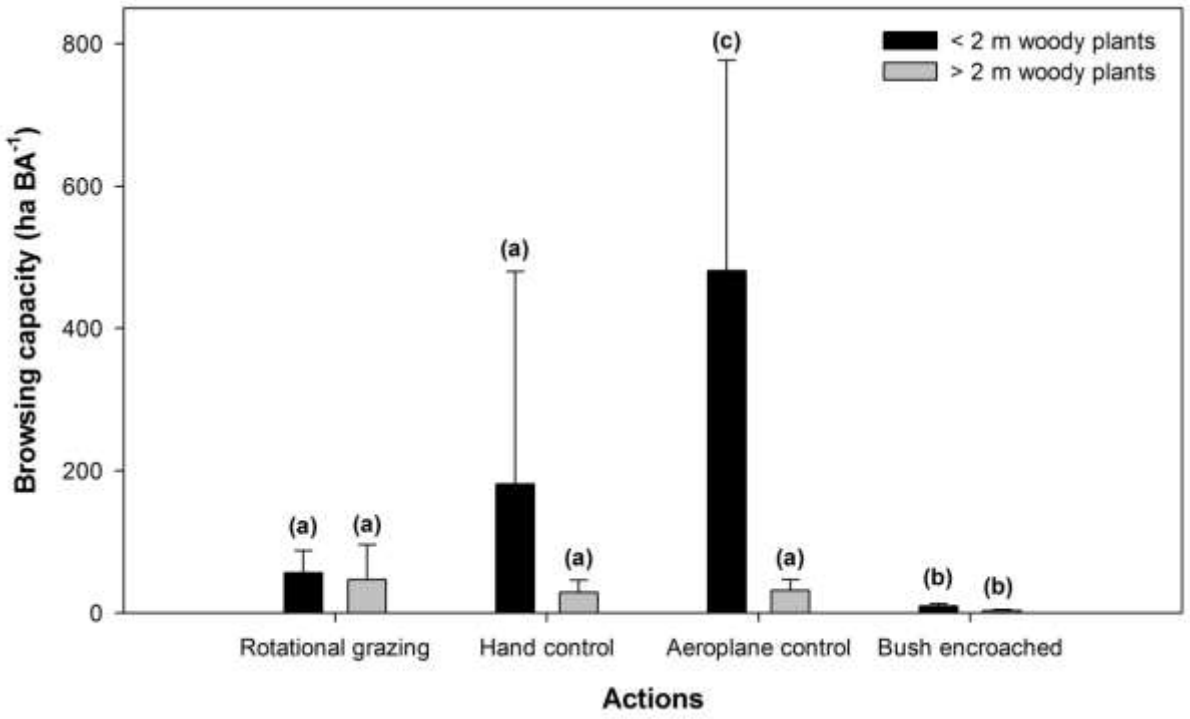

Figure 5.5: Effect of rotational grazing (n = 8), chemical hand control (n = 7) and chemical aeroplane control (n = 7) actions compared to bush-thickened sites (n = 15) on the browsing capacity (ha BU^{-1}): Bars indicate mean values with standard deviations. Different lower-case letters indicate a significant difference at $p < 0.05$ (ANOVA with post-hoc Tukey's HSD test).

5.3.7 Relationships between the status of the woody and grass layer

A negative relationship ($r^2 = 0.85$; $p < 0.001$) was found between the woody phytomass and grass forage production (Figure 5.6a). The relationship between woody phytomass and grazing capacity was highly significant ($r^2 = 0.84$; $p < 0.001$) (Figure 5.6b).

The results presented in Figures 5.6a and b were complementary to those found in sections 5.3.6.1 and 5.3.6.2. In addition, results from the linear regression analysis confirm the findings of low grass forage productions and grazing capacities in the high woody plant phytomass densities of the BT sites. However, the lower woody densities sites under RGM, HC and AC had higher grass forage productions and grazing capacities.

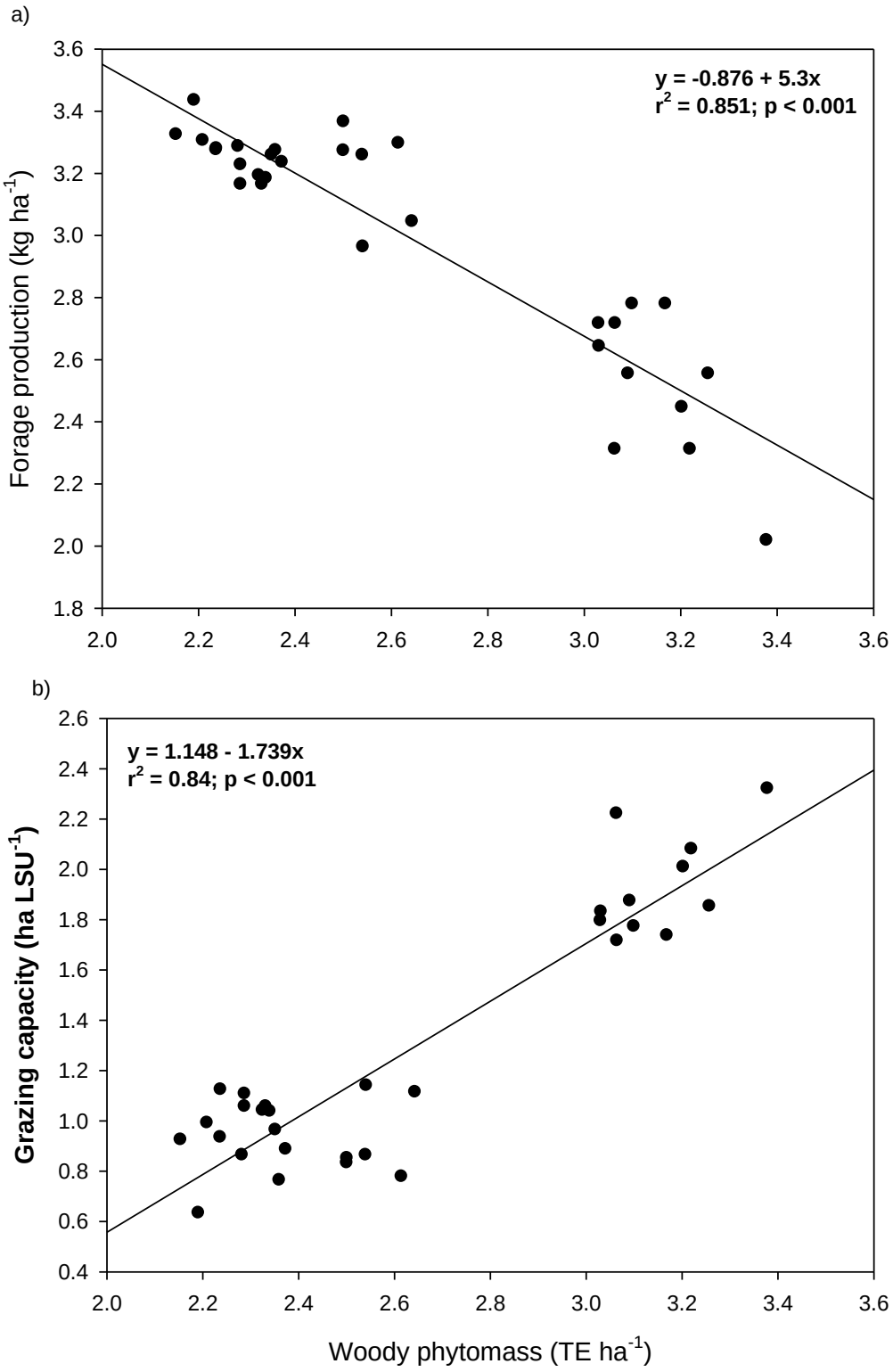

Figure 5.6: a) Relationship between woody phytomass and forage production, and b) relationship between woody phytomass and grazing capacity based on data from all (n = 37) sampling sites across the actions: Data was log-transformed and fitted with simple linear regression.

5.3 Discussion

5.3.1 General patterns in altered plant species composition

Result from the Principal Component Analysis (PCA) ordination indicated that the management (RGM) and restoration (HC and AC) actions accounted significantly for most of the grass and the woody compositional patterns. Clear confirmation of the pre-defined restoration and management action groupings was found for the different treatments. With a probability of the HC actions with the highest species composition similar to that of the "benchmark" RGM sites which are characterised by more palatable grass and woody species such as *Schmidtia pappophoroides* and *Acacia erioloba* respectively.

GFG 1 was also found to be more dominant in the RGM and HC sites, with species (i.e. perennial, more palatable, climax grass type species such as *S. pappophoroides, Brachiaria nigropedata and Anthephora pubescence*) that are largely grazing sensitive. Care must be taken not to overgraze these species because of their value as grazing species, being highly preferred by grazing herbivores. Due to the high forage production of these species, they usually also indicate rangelands in a good or better condition (Van Oudtshoorn, 2012). In case of overgrazing or as a result of bush thickening, these species will begin to decline, and the area will then start to be dominated by pioneer and sub-climax type species better adapted at tolerating a higher grazing intensity (Donaldson & Kelk, 1970; Coppock, 1994; Angassa, 2005).

In the selective HC sites, the density of the palatable woody plant species, such as *G. flava,* was found to be the highest. *G. flava* is not a targeted species for removal by way of selective hand chemical control, and this species' high competitiveness for soil moisture, given its shallow root system as indicated by Skarpe (1990[a,b]), may well explain its high dominance in the HC sites. The results calculated with the aid of ANOSIM/SIMPER and PCA highlighted the high similarity between RGM and HC sites regarding vegetation composition. Consequently, it can be assumed that HC might be a more effective action to restore species composition when compared to the RGM "benchmark" site.

Taking into account the results from the species frequency distributions, ANOSIM/SIMPER and the PCA, the BT sites are clearly dissimilar from the RGM sites and the two chemical control actions as far as species composition is concerned. This results in a significant higher occurrence of annual, less palatable grass species (such as *Schmidtia kalahariensis)* and encroacher woody plant species (e.g. *Acacia mellifera),* both being commonly associated with disturbed areas and regarded as indicative of poor rangeland conditions (Van Oudtshoorn, 2012).

The results, however, indicate that perennial, more productive grass species, such as *S. pappophoroides, Aristida stipitata* and *Eragrostis lehmanniana,* have not disappeared from the BT areas all together and, therefore, assumingly also not from the soil seed bank. This is confirmed by results from a study conducted by Angassa (2005) which indicated that palatable climax species (e.g. *Cenchrus ciliaris*) do, in fact, occur within BT areas, but that less palatable pioneer and sub-climax grasses are bound to increase over time.

BT areas were dominated by various GFGs, ranging from GFG 2 to 5. These groups are mostly characterised by pioneer to sub-climax species that range from perennial to annual species and are grazing tolerant. Although the grass species *Aristida stipitata* was the second most frequent occurring grass species in the BT sites, it is rarely preferred by animals for grazing (Van Oudtshoorn, 2012). Such unpalatable species mostly display traits such as an advanced seed-dispersal strategy and a short lifespan, which could be traits that have been developed to endure high grazing pressure and moisture stress (Morris *et al*., 1992; Palmer *et al*., 2003; Hester *et al*., 2006; Van Oudtshoorn, 2012). The vegetation dynamics and natural successional processes of rangelands can be influenced positively by the high occurrence of annual, unpalatable pioneer-sub-climax grasses that are grazing tolerant at the start of the succession process (Van den Berg & Kellner, 2005). They also provide shade and cover and increase the soil moisture content for the establishment of climax grasses (Van Oudtshoorn, 2012).

In terms of this study, Skarpe (1990[a]) also indicated that the two woody species responsible for the thickening of woody species in the rangelands and, therefore, BT areas were found to be *Acacia mellifera* and *Grewia flava*. These two species have a

mostly lateral root system, affording these plants an advantage in terms of water uptake above other woody species following heavy grazing or low grass layer cover (Skarpe, 1990[a]). Other studies also indicated that *A. mellifera* with its extensive lateral root systems as well as other encroacher species such as *Dichrostachys cinerea* and *Terminalia sericea* are dominant in very high bush-thickened states and are able to suppress the growth and production potential of grass plants (Scholes & Archer, 1997; Richter *et al*., 2001; Smit, 2003[b]; Hagos & Smit, 2005; Riginos & Young, 2007; Riginos *et al*., 2009). This may explain why these two woody species are dominant in the BT areas where the grass cover is low. Thus, with an increase in grazing pressure and competition from woody plants in the BT areas, the assumption can be made that the vegetation is in a stage of transition towards less desirable pioneer grass species only. Consequently, there is a need to monitor rangelands over a longer period of time in order to determine changes in species composition and, in this way, to respond immediately in an attempt to conserve valuable climax grasses (Angassa, 2005).

The AC sites, being a non-selective approach, form their own clusters which were found to be more dissimilar when compared to RGM mainly due to an abundance of *Boscia albitrunca* and a lower dominance of *Schmidtia pappophoroides*. The woody plant *B. albitrunca* has a deeper root system (up to 68 m, see Canadell *et al*. 1996) compared to some of the other woody plants, such as *A. mellifera* and *G. flava.* This deeper root system could well be assumed to be the reason why *B. albitrunca* is less affected by the arboricide used for chemical control under the recommended dosages. This assumption is also reflected in the PCA ordination, namely a relatively high frequency distribution of *B. albitrunca* in the AC sites. Nevertheless, *B. albitrunca* can be affected by the arboricide over many years as the arboricide percolates deeper into the soil profile. However, research on how *B. albitrunca* is affected by the arboricide has yet to be carried out. *Acacia- , Grewia-* and *Dichrostachys* species have a mostly lateral root system, which can sometimes extend to 10 or more times the height of the shrub or tree (Skarpe, 1990[a]; Smith & Rethman, 1998; Smit, 2003[b]; Joubert *et al*., 2008; Sea & Hanan, 2012). Such a lateral root system could result in a higher and quicker uptake of the arboricide when chemically controlled.

There was a profound shift from the dominance of GFG 1 in the RGM and HC sites towards a significant higher dominance of GFGs 2 (perennial, palatable, grazing tolerant, pioneer-sub-climax grass) and 4 (weak perennial palatable species) in the AC sites. The dominance of these GFGs with only medium grazing values resulted in the lower grazing capacity reflected in section 5.3.6.1. GFG 2 and 4 consisted of only pioneer to sub-climax type species with smaller tuft formations. These grasses are able to provide the necessary shade and soil cover for climax-type species to establish with the implementation of AC actions under the correct grazing intensity and management (Van Oudtshoorn, 2012).

The total removal of woody plants (such as during the non-selective AC actions) has had a negative effect on the grass plant species associated with tree canopies, e.g. species such as *Panicum maximum* (Smit & Swart, 1994; Smit, 2005[b]). A rapid increase in less palatable grasses at the expense of species that are commonly found in shady environments, such as *P. maximum* in totally cleared areas, will result in the reduction of the palatability of the grass layer. Mature woody plants provide sub-habitats that are favourable for the establishment and growth of many grass plants (Belsky *et al*., 1989; Belsky, 1994; Smit *et al.* 1996; Smit, 2004; Hagos & Smit, 2005). However, certain perennial grass plant species will increase only to a certain extent with an increase in the woody plant densities, as the competition between the woody and grass layer for soil moisture and nutrients becomes too intense, causing a decline in the grasses (Seghieri *et al.,* 1994; Smit & Swart, 1994).

Interestingly, GFGs 2 and 4 were both found to be abundant in AC and BT sites (i.e. the two actions with the lowest and highest woody densities). This may be an indication that these two GFGs are well adapted to occur and dominate in both open grasslands as well as very dense woody areas. This phenomenon could be due to the fact that the species in these two GFGs are grazing-tolerant, pioneer to sub-climax grasses which are better adapted to establish in disturbed areas, such as BT sites, where more open, bare soil patches occur. The assumption is also that these species have previously dominated in BT sites and have been able to build up a soil seed bank. The latter explains why, after the removal of the encroacher woody plants through AC, dominant seed germination would be that of pioneer to sub-climax type grasses.

Previous studies have shown that bush thickening results in marked changes in the grass species composition (Donaldson & Kelk, 1970; Coppock, 1994; Angassa, 2005). An increase in annual, less palatable pioneer grasses (e.g. GFGs 5 and 6) could be an indication that management actions need to be adapted to lower grazing pressures, or that restoration actions need to be applied to increase the cover and abundance of more climax-type grass species. The local extinction of key GFGs (such as GFG 1) will ultimately lead to the surpassing of the threshold, and the ecosystem will not be able to revert towards the benchmark state indicative of a better rangeland condition, unless active restoration and better management actions (e.g. chemical shrub control in combination with sustainable grazing management actions) are implemented (Bosch & Kellner, 1991; Beisner *et al.*, 2003; Briske *et al.,* 2003; Stringham *et al*., 2003; Suding *et al*., 2004; Briske *et al.,* 2006; Hobbs *et al*., 2009; Suding & Hobbs, 2009).

In terms of effectiveness and the choice of management protocols, future attempts to restore rangeland ecosystems ought to take cognisance of these changes in the composition of grass species seeing that such a composition may well impact the vegetation's successional dynamics following on bush control attempts (Smit, 2004; Angassa, 2005). However, from the results of this study, the assumption can be made that the remaining populations of palatable grasses in the BT sites indicated a certain regeneration potential towards better conditions following on bush control. This assumption is supported by Martin and Morton (1993) who reported a greater density of perennial grasses in sites treated for mesquite (*Prosopis velutina*) three years earlier compared to sites left untouched. Reduced moisture competition following on the removal of competitive woody plants can, in turn, increase the success of alternative restoration measures such as the (re-)introduction of desired climax grass species through re-vegetation or other investments to improve rangeland quality from both an ecological and an economic perspective (Wolfson & Tainton, 1999; Smit & Rhethman, 2000; Smit, 2005[a]; Lohmann *et al*., 2012).

5.3.2 Effect of actions on density and productivity parameters

Forage production and grazing capacity were found to be significantly higher in the chemical bush control sites (HC or AC) when compared to the BT sites which were not controlled. The bare soil was colonized by grasses in the areas with a lower

woody plant density. This may be due to the chemicals used during the control methods, with a decrease in the runoff of rainfall and an increase in the available soil water needed for the establishment of grasses (Smit & Rethman, 2000). As already mentioned with regard to the results derived from the similarities in species composition, if these bare areas with former high woody plant densities could be colonized by grasses after the implementation of chemical actions, it may imply that a suitable soil seed bank still existed in these BT areas, albeit low in numbers.

In *Acacia* species savanna veld types, an absolute increase in grass forage production of between 300 kg ha^{-1} and 2500 kg ha^{-1} was estimated after the clearing of competitive woody plants (Scholes, 1987). In the present study, a 1076.4 kg ha^{-1} (HC) and 1406.1 kg ha^{-1} (AC) difference in the grass forage production was obtained in the controlled sites when compared to the BT sites. This is similar to previous results that show an increase in grass forage production if woody plants are thinned out in BT areas (Dye & Spear, 1982; Moore *et al*., 1985; Smit, 1994, Richter *et al*., 2001; Smit, 2005). According to Smit *et al.* (1999), bush thickening causes a huge decline in the grazing capacity of rangelands in various areas of southern African savannas. With a decrease in grass forage production, the grazing capacity decreases accordingly, thereby reducing the rangeland's overall productivity. In the present study, it is assumed that the higher grass forage production found in chemically controlled sites can mainly be ascribed to the reduced competition between grasses and woody plants for water and nutrients. This may also lead to an increase in the regeneration potential from the soil seed bank after four years of chemical applications in both HC and AC sites.

The HC sites had a lower grass forage production compared to the RGM and AC sites, but due to the high average frequency of palatable grass species, the HC sites had the second highest grazing capacity compared to the other actions. Rotational grazing is also practised in the HC areas, which contributes to the fact that the vegetation can rest for longer periods and can, therefore, recover easily. This may be the reason for the higher grass forage production found in these sites. Other studies have also shown that a resting period during RGM systems accelerates the improvement of rangeland condition (Müller *et al*., 2007; Brunson & Burrit, 2009) and allows a higher total stocking rate within a certain time period (Quirk, 2002). The suppressive effect of the remaining woody plants not controlled in the HC action

could still inhibit grass growth, which may lead to lower grass forage production compared to the AC sites. During the AC, most of the woody competition exerted on the grass layer was removed, which may be the reason for the higher grass forage production compared to HC and BT sites. It is thus important to take into account the total woody plants removed during HC actions in order to determine whether it will be effective enough to decrease competition between woody and grass plants to such an extent that it will favour the primary grass phytomass production and the establishment of climax grass species.

As mentioned before, from the data found in the ANOSIM/SIMPER and PCA ordination, *Acacia mellifera* was found to be highly correlated with BT sites. Data published on the effect of *A. mellifera* densities on the grass forage production highlighted that the grass forage production (119 kg ha^{-1}) decreased as the number of mature woody plants per unit areas increased (1071 woody plants ha^{-1}) (Donaldson & Kelk, 1970). In the absence of woody species, the grass forage production was 1071 kg ha^{-1} (Donaldson & Kelk, 1970). Although the study was based on a single season, the results indicated differences in the grass forage production between BT and chemical control actions.

The results from the linear regression of the present study have shown an exponential negative relationship between the woody phytomass and grass forage production. The results further indicated an 81.5% decrease in the grass forage production at woody plant densities of 1445 TE ha^{-1}. The potential woody plant competitiveness at high densities is seen as an important determinant of the total grass forage production (Smit & Swart, 1994). This linear decrease in grass forage production is in accordance with other findings (e.g. Donaldson & Kelk, 1970; Harrington & Johns, 1990; Moore *et al*., 1985; Moore & Odendaal, 1987; Smit & Swart, 1994, Smit & Rethman, 1999; Richter *et al*., 2001) in savanna-type vegetation. Such an intense suppression of grass growth may result in livestock and game farms no longer being economically viable (Smit *et al*., 1996; Smit & Rethman, 1999; Richter *et al*., 2001; Smit, 2004).

A significant negative correlation was found in the same study area with an 82% decline in grass forage production at woody plant densities of 2500 TE ha^{-1} (Richter *et al*., 2001). Another study has found no reduction in forage production up to a

density of 200 TE ha^{-1}; however, the grass forage production declined linearly with an increase in the woody plant densities above 200 TE ha^{-1} (Moore & Odendaal, 1987). At a woody plant density of 2000 TE ha^{-1}, almost all grass growth was suppressed (Moore & Odendaal, 1987) and can, therefore, undoubtedly lead to a pseudo-drought effect as a result of the woody-grass competition. Richter *et al*. (2001) ascribed this decrease in grazing capacity to the high incidence of *Acacia mellifera* in the Molopo Bushveld vegetation type, stating that this woody species has a greater negative effect on the grazing capacity and forage production of rangelands than other woody species such as *Terminalia sericea*.

In the AC sites, a significantly lower browsing capacity was obtained if compared to the BT sites. This was expected, as most of the woody plants in the AC areas were killed by the non-selective approach, with the exception of the *Boscia albitrunca* species. The BT sites had the highest browsing capacities with less land needed to sustain one browser unit due to the high dominance of *A. mellifera* and *G. flava,* which are both palatable species. Although the browsing capacities for the BT sites may be very high, these areas are not always accessible to livestock and game as a result of the high densities of the woody plants (Skarpe, 1990[a]; Twyman *et al*., 2002; Dougill & Thomas, 2004).

Results suggest that RGM and HC actions are relatively successful in sustaining high grass productivity by suppressing high woody plant densities. Previous studies have shown that a high grass growth rate and grass phytomass productions are able to suppress seed germination and reduce the demographic transition probability of woody plants, thereby reducing woody seedling survival and establishment due to competition for moisture and other resources (Riginos, 2009; Hagenah *et al*., 2009; Kgosikoma *et al*., 2012). Both RGM and HC sites were found to have a less dense woody layer and high grass forage productions of mainly climax grass species, which are assumed to suppress the establishment of woody plant seedlings. These two actions (i.e. RGM and HC) were found to have the highest grazing capacity and forage production, the two productivity parameters cattle farmers are most interested in to increase the production rate, animal condition and profit.

Although the control of the encroacher species resulted in a less dense woody cover layer in the selectively hand-controlled sites, a similar woody composition than that

of the RGM sites was still observed, which could be utilised by browser animals to improve rangeland productivity and profit. Monitoring and management strategies should recognise this pattern, as it is very effective to still maintain a high grazing capacity and forage production while decreasing the grass-woody ratio typically found in savanna ecosystems. However, if left unmanaged and untreated without any follow-up chemical treatment, the density, leaf biomass and average individual height of the woody layer within the two chemical control actions are expected to increase over time (Barac *et al*., 2004; De Klerk, 2004; Smit, 2005). The woody layer is a vital component of semi-arid savanna ecosystems and should be managed within the framework of economic viability and ecological responsibility (Richter *et al*., 2001; Smit, 2004; Dalgleish *et al*., 2012).

Richter *et al*. (2001) recommend that the high densities of encroacher woody plant species in the Molopo region be controlled, preferably by making use of chemical arboricides, as this results in a significant increase in grass forage production. However, thinning of woody plants will result in immediate changes in the woody-grass competition which will ultimately determine the growth and structure of savanna systems (Smit *et al*. 1996). It is commonly acknowledged that when mature woody plants capable of suppressing saplings are lost from the savanna ecosystem, more frequent and repeated efforts will be required to prevent re-thickening by woody plants (Smit, 2004; 2005). Cleared areas need to be protected against intense utilisation from herbivores until a stable grass layer has been established. However, despite the known disadvantages of total woody plant clearing, non-selective AC is still often practiced in the Molopo region.

5.3.3 Is hand control better than aeroplane control?

Based on the results described above, the selective HC method, together with a sustainable RGM, may provide the best possible way to control bush thickening and to restore the rangelands to more productive and stable savannas. The main reasons include the following: a) A significantly higher grazing and browsing capacity was found for HC with a more balanced grass-woody ratio compared to AC. b) HC may be more expensive and require more labour to implement in high woody plant densities, but a higher abundance of valuable grass and woody species (i.e. key natural resources such as the pods from woody plants for utilisation by both grazer

and browser animals) was gained compared to the non-selective AC. c) A more structured savanna with different height classes and especially larger woody plants is developed. Larger woody plants are able to act as water pumps, create fertile islands for soil enrichment and suppress the establishment of new woody plant seedlings, thereby creating more stable environments for grass establishment (Smit & Rethman, 2000; Smit, 2004; Hagos & Smit, 2005). d) HC creates more open grasslands with a scattered woody layer typical of a savanna ecosystem, which improves the esthetical value and increases wildlife visibility for hunting and eco-tourism.

The HC method, together with a sustainable RGM system, will also be able to improve the rangeland's forage productivity (especially if browser animals are inter-mixed with grazer animals) and decrease the degradation of the rangeland.

As the initial conditions at the sites and long-term implications of using arboricides as a chemical control method are not known, it may be argued that over time, the species composition of areas were AC was applied may return to that of HC and RGM. The latter will require long-term assessments which are not included in the present study. The sequence of succession expected after the removal of most of the woody plants with AC that are characterised as encroacher species, such as *Acacia mellifera,* may be the first to colonize the controlled areas and can then be regarded as "pioneer" type species. Depending on the dosage used, *Tebuthiuron* residues in the soil (the active ingredient used in the arboricide during chemical control) may preclude the re-colonization of seed by woody species for at least eight years following the application of the arboricide (Du Toit & Sekwadi, 2012). This can provide the opportunity for pioneer, sub-climax grasses to establish. These pioneer, sub-climax grasses would provide better environmental conditions for the establishment of climax-type grasses over the long term if proper management (correct stocking densities) is applied. It may also be possible that before the arboricides has percolated out of the top soil layer, the grass layer would have been established in a climax condition with proper biomass material, which may suppress woody plant establishment.

Another aspect to consider may be that the grass populations in the HC sites were in a better condition than those sites where AC was applied (i.e. lower woody plant

densities in HC sites and, therefore, less competition on the grass layer from the woody plants). Furthermore, the condition of rangelands surrounding chemical cleared sites may have a higher seed input by dispersal into the controlled sites, which may lead to faster regeneration. All these factors may contribute to the current differences between AC and HC sites. AC sites are also still surrounded by higher densities of woody species (i.e. bush thickened blocks), which may act as physical barriers for wind-dispersed grass seeds.

5.4 Conclusions

The results on the GFGs and from the PCA and ANOSIM/ SIMPER analysis clearly demonstrated the existence of differences in the grass- and woody compositional patterns with regard to the different restoration (HC and AC) and management (RGM) actions compared to BT sites. It is clear that the grass layer in BT areas is in a very poor condition compared to the other three actions. RGM and both chemical control actions proved to be effective and to significantly increase the grass forage production, together with improved establishment of more perennial, palatable, climax grass species and, accordingly, a significantly improved grazing capacity.

A selective approach of woody plant clearing (such as HC) may be a more appropriate and sustainable solution and will ensure a more balanced grass-woody ratio that will take into account the structure and size of the woody plants, as well as the type of species that need to be thinned out. Results from this study also show that patterns of grass species, GFGs and woody densities indicated that RGM and HC are more effective and resemble a productive and stable savanna system, providing important key resources for both grazing and browsing herbivores.

The extent and negative effect bush thickening has on the decrease in grass forage production and grazing capacity (i.e. the rangeland’s productivity potential) and other natural resources is clear, and for this reason, serious consideration ought to be given to the mitigation of bush thickening in the Molopo region and other savanna systems that are highly susceptible to woody plant thickening.

6

Conclusions and recommendations

6.1 Conclusions

6.1.1 Integrated assessment of restoration and management actions

The first research objective of this study was to test the viability of the IAPro protocol as an integrative, bottom-up approach to evaluate the suitability of locally applied restoration and management actions for the mitigation of rangeland degradation in the identified study area, namely Molopo. This objective was achieved by first utilising a chain-referral process to identify participants who could constitute a suitable multi-stakeholder platform (MSP) as per Step 1 in said protocol. As per Step 2, semi-structured interviews were then conducted with the identified participants to identify indicators and locally applied restoration and management actions before involving the same participants in ranking the thus identified, shortlisted indicators and actions by assigning each a relative weight value (Step 3). This was followed by the quantification of the bio-physical indicators with the aid of vegetation sampling methods whilst simultaneously assessing the socio-economic indicators with the aid of a questionnaire (Step 4). In Step 5, the resultant indicator rankings and biophysical assessments were then combined in a multi-criteria decision analysis where after the IAPro assessment process was concluded, resulting in a final integrative assessment (Step 6).

The 46 participants identified for this study all participated successfully in the six steps outlined above. As a result, a total of 31 site-specific indicators could be identified and subsequently shortlisted as 11 indicators to which the participants had assigned a relative value.

Following on the weighting exercise, the resultant six (6) bio-physical indicators were quantified in the field and the five (5) socio-economic indicators qualitatively ranked

with the aid of a questionnaire administered in the course of a workshop attended by most of the participants. The relative weight values of the 11 indicators were combined in a multi-criteria decision analysis (MCDA) which ranked the alternative restoration and management actions according to their relevancy and performance. This outcome was then shared with the members of the MSP in order to stimulate discussion among the land users and to solicit further inputs. Following on controlled implementation, certain restoration and management actions were also evaluated in an attempt to aid land users' decision making as far as the sustainable management of their rangelands is concerned.

The overall response of those forming part of the MSP was positive, and the stakeholders (SHs) were definitely willing to participate and learn from the IAPro. In particular, the identification of the site-specific indicators relevant to participants in the Molopo area, combined with the subsequent quantification of these indicators, proved to be of great practical value for the land users and even a necessity in as far as making informed decisions on the future sustainable management of their rangeland is concerned.

With the aid of IAPro, qualitative insights derived from the participatory research could be combined with the quantitative results gained by way of biophysical assessments, the end results being far more accurate and relevant than the results either approach used in isolation would have been able to achieve. The multitude of indicators mentioned by the local stakeholders is indicative of the wealth of indigenous knowledge and information available amongst SHs in the Molopo region.

The exercises to weight indicators, the group discussions conducted during the two workshops and the completion of collective integrated assessments contributed towards a more diversified understanding of the environment based on local traditional and scientific expertise. In this way, the gap between the local SHs' indigenous knowledge and scientific quantitative results could be addressed, resulting in land users being better informed and enabling them to adapt to changing environmental conditions and to observe signs of rangeland deterioration more effectively. The perceptions and objectives of the land users regarding rangeland management had a direct influence on the outcomes of the integrated assessment, which could have contributed to participants' overall acceptance of the results.

Results of this study show that the technical implementation of restoration and management actions will be highly dependent upon the land-tenure types and management objectives under consideration in the Molopo region. Undisputedly, the tested approach holds direct benefits for land users under various land-tenure systems. However, in communal farming systems, a sustainable rangeland management system, together with effective bush control actions, is often difficult to implement due to inappropriate governance structures, strong competition for natural resources and the high associated costs of materials, such as fencing and chemicals, for bush control.

From this study, it can thus be concluded that participatory research can promote sustainable rangeland management in southern Africa by actively involving local land users in the evaluation, decision-making and execution processes. Results showed that this type of participatory assessment holds great promise to help identify the best and most sustainable actions to mitigate rangeland degradation and to improve rangelands' condition and production potential.

6.1.2 Impact of chemical control actions on the vegetation of the Molopo Bushveld

Rangeland degradation caused by bush thickening and the possible effect thereof on rangelands' production potential are of great concern to the land users in the Molopo region. For this reason, the second research objective in this study focused on the evaluation of the impacts of two chemical bush control actions (hand and aeroplane) on changes in the vegetative composition, grass phytomass production and woody plant densities.

The findings from the PCA and ANOSIM/SIMPER analysis clearly revealed that the respective restoration and management actions (HC, AC and RGM) do result in differences in grass and woody compositional patterns when compared to the bush-thickened (BT) sites that were used as a control. Rotational grazing management (RGM) and hand-controlled (HC) sites had very similar woody plant densities and a similar grass layer composition, with the mostly perennial, palatable, climax grasses present being characteristic of a good condition. Aeroplane control (AC), which is a highly unselective control method, resulted in the loss of important ecological features, such as large woody plants, that can provide more sustainable

environments (i.e. the requisite nutrients, moisture and shade) for the establishment of perennial grasses. Consequently, the grass species composition of AC sites were found to be in a poorer condition with more annual, pioneer to sub-climax-type species compared to RGM and HC sites.

BT sites were characterised by grass- and woody plant species mostly associated with disturbed and overgrazed rangelands. The grass layer of the BT sites was also characterised by a higher dominance of pioneer to sub-climax grasses compared to RGM and HC sites.

The effect chemical control (HC and AC sites) vs. no control (BT sites) had on the productivity parameters of the vegetation was also assessed. Results indicated that the rangeland condition (i.e. grass forage production and grazing capacity) in the RGM and both chemical control sites was better compared to the BT sites. However, the HC and AC actions resulted in a significant decrease in the number of woody plants. With the reduction in woody plants, the competition with the grass layer was also decreased, which resulted in higher grass forage production and grazing capacity values in the HC and AC sites. BT sites, where the competition rates between woody and grass plants for moisture was still high, had significantly lower grass forage productions and grazing capacity compared to RGM, HC and AC sites.

Results also revealed a negative relationship between the woody phytomass and grass forage production. A high woody plant density is regarded as an important competitor when determining the total grass forage production. A negative relationship was also found between the woody phytomass and grazing capacity, since the woody phytomass increased concomitantly with a decrease in grazing capacity.

The assumption is that the high grass forage production found in the RGM sites suppressed the establishment of woody seedlings, thereby maintaining a relevant sparse woody density. The low woody plant densities of the RGM and both chemical control (HC and AC) sites resulted in very low browsing capacities. As expected, the BT sites had a significantly higher browsing capacity for both woody height classes (i.e. < and > 2 m) when compared to RGM and the two chemically controlled sites.

Soil enrichment by woody plants is a very slow process. Significant changes in soil properties as a result of the implementation of chemical control and RGM actions could, therefore, not be established in the relatively short lifespan of this project. However, since changes can be detected over time, it is recommended that soil properties be monitored over the long term to determine whether restoration or management actions have any effect on the pH and organic carbon of the soil.

Although the effect of bush thickening control could be detrimental to the environment, it can increase the grass forage production and grazing capacity. Therefore, serious consideration of these control methods in the Molopo region and other savanna systems susceptible to woody plant thickening to increase rangeland productivity potential is warranted.

All factors considered, a selective woody plant control approach (such as HC) may be a more appropriate and sustainable solution compared to AC, especially if the objective is to increase overall forage production and grass species composition for grazing animals. This will contribute to a more balanced grass-woody ratio and also establish a better structure in the height classes of woody plant species. In sum, results from this study show that the patterns and compositions of grass species, GFGs and woody densities indicated by RGM and chemical HC best resemble a productive and stable savanna system that provides important key resources to support both grazing and browsing herbivores.

6.2 Recommendations

6.2.1 Recommendations on the IAPro (Integrated Assessment Protocol)

Some difficulties were observed during the indictor ranking exercise (steps 3a and 3b), as certain SHs did not quite understand or had different perceptions on what was meant by some of the terms and indicators used, such as “biodiversity” and “rangeland degradation”. This problem became evident when land users from different land-tenure systems and cultural backgrounds were grouped together, especially regarding the ranking of the biophysical and socio-economic indicators. The recommendation, therefore, is that the results for SHs from the different land-tenure groups be analysed separately during the integrated assessment process,

since the contexts of people from, for example, commercial and subsistence (communal) management systems differ quite significantly (Reed *et al*., 2006; Von Maltitz, 2009, Dreber *et al*., 2013).

It should be kept in mind that no absolute “best” restoration or rangeland management action, as described by the IAPro, exists since the evaluation of the actions depends on tradeoffs between land-user perspectives and farming objectives as well as the dynamic socio-economic contexts which differ among commercial and subsistence farmers. These factors ought to be considered and must be adapted from case to case (Reed *et al*., 2006; Von Maltitz, 2009; Bautista & Orr, 2011; Dreber *et al*., 2013). Likewise, this emphasises the need to carefully consider how the knowledge and results are transferred to the participants and to ensure that the information is correctly understood by the respective SHs. Thus the recommendation is that the stakeholder platform be realigned by stratifying the land users involved according to each specific socio-economic setting and its objectives. This would undoubtedly enable the improved identification of more suitable and affordable actions based on an enhanced problem-specific contextualisation (Dreber *et al.*, 2013).

In following the IAPro, it was sometimes difficult to identify enough sampling sites suitable for the replication of certain actions and to reduce variability in the data. The latter was caused by the heterogeneity in the biophysical environment and differences in the design of restoration and management strategies applied. Although it is very difficult, often very time consuming and requires a good knowledge of the study area, especially in the selection of sites in the natural environment, it is recommended that a representative number of replicates be carried out to ensure sound statistical evaluation of the results (Dreber *et al*., 2013).

Another challenge for the participants of the MSP was that most land users had to travel vast distances to attend the workshops. This resulted in varying participation rates for land users, which may be problematic when attempting to ensure sound interpretation of the results. To address this challenge, it is recommended that both iterations (step 3a & 3b) be held with the same group in a single meeting (Bautista & Orr, 2011).

Research on rangeland degradation and sustainable restoration and management actions should continue, especially following a participatory approach in South Africa's communal areas (Hoffman & Ashwell, 2001). However, sustainable rangeland management actions should also be supported and maintained in commercial farming areas, as this farming sector is crucial for the country's productive future (Hoffman & Ashwell, 2001). As bush thickening is a huge problem in both communal and commercial rangelands in the Molopo, it is recommended that rangeland degradation be combated at the local level, and that as Von Maltitz (2009) suggested, institutional mechanisms and structures be instituted to facilitate sustainable rangeland management amongst the different forms of land use and ownership.

Dreber *et al*. (2013) recommended that replications and comparisons of the PRACTICE approach be executed in other areas as well, in order to evaluate this participatory approach's suitability and applicability in similar savanna regions. In South Africa, environmentally and socio-economically accepted restoration and management actions are of importance to combat rangeland degradation, which may imply that an adapted PRACTICE approach could be a promising tool for the exploration of alternative restoration and management actions not identified by the local members of the MSP (Dreber *et al.,* 2013).

The work by Reed *et al.* (2007, 2008) provides a monitoring tool for rangeland degradation assessments, and the possibility of combining this approach with the PRACTICE approach ought to be explored. By combining the two frameworks/approaches, it might be possible to create a more holistic framework in order to improve land users' decision making and to ensure a more sustainable rangeland management system underpinned by a long-term monitoring programme (Dreber *et al.,* 2013).

6.2.2 Recommendations on chemical bush control actions

Generally, the recovery of disturbed areas in arid and semi-arid environments is a highly protracted process taking up to several decades, especially if no active restoration or management actions are implemented (Bradshaw, 1994 *in* Van Rooyen, 2000; Snyman, 2003). In this study, the mitigation of high woody plant densities (i.e. bush-thickened areas) by chemical control methods have revealed that

it can be advantageous when applied in degraded, highly bush-thickened areas, especially to increase grass forage production and to improve the grass species composition towards more perennial, climax grasses. It is recommended that the positive as well as negative ecological impacts of bush control actions to improve rangeland productivity be demonstrated effectively to and discussed with land users (Angassa *et al.*, 2012) (e.g. the longevity of Tebuthiuron residues remaining active in the soil of semi-arid grasslands as discussed by Du Toit and Sekwadi (2012) and the decrease in palatable large trees that affects the aesthetic value of the environment). The chemical clearing of bush-thickened areas is a long-term process, and not all chemical approaches may be ecologically sustainable, as ecosystems are all in different transitional states, which can influence the effectiveness of bush control actions (Meyer *et al*., 2007; 2009).

Certain woody plants, such as *Dichrostachys cinerea,* require a very high dosage of arboricide in order to be controlled, compared to the other woody plants. Likewise, certain tree species, such as *Acacia erioloba,* are also more sensitive to arboricides and may be killed if high dosages are used or the control of other species is not targeted. The recommendation, therefore, is that those people applying the arboricide by hand must be able to distinguish between the different woody plant species in order to ensure effective control of only certain problem species responsible for bush thickening, such as *A. mellifera* and *D. cinerea*.

Usually, it is very expensive to implement chemical control methods, whether by hand or by aeroplane. It is, therefore, recommended that any method should only be implemented initially on a small scale and be considered later for larger areas and in a follow-up stage should re-thickening occur. According to Trollope *et al*. (1989), it may be necessary to resort to chemical control methods in cases where (a) very high woody plant densities exist and the accumulation of grass biomass (fuel) is insufficient to support a fire hot enough to kill the top-growth of the targeted woody plants, (b) the majority of the woody plants have grown to a height beyond the reach of browsing animals, (c) animal movement is restricted due to high woody plant densities, (d) the woody plant species are largely unpalatable to the animals and (e) the arboricide being used mainly targets problem woody plant species and not the more palatable woody plants.

The removal of most or all of the mature woody plants can be effective in a much shorter time when using AC, but if mature woody plants capable of suppressing saplings are lost from the system, the treated area could deteriorate to a more thickened state (i.e. an unstable system), thus requiring more frequent and repeated efforts to prevent re-thickening by woody plants over the long term (Smit, 2004; 2005[a]; Hagos & Smit, 2005). Hence, the recommendation is that AC areas be protected against intense utilisation by herbivores until a stable grass layer has been established; otherwise, if herbivore numbers are not correctly managed, an increase in grass forage production and grazing capacity may not be achievable.

Smit *et al.* (1996) suggested the following be kept in mind when combating bush thickening in savanna ecosystems: (a) Woody plants are a natural component of savanna systems, and the different floristic components thereof have evolved over many years. (b) Woody plants are essential to the savanna systems in terms of soil enrichment, shade, burnable fuel, medicine and aesthetic value. Taking the afore-going into consideration, using AC could, therefore, be a drastic intervention from an ecological perspective. Therefore, in the Molopo study area, a sustainable RGM system with a low stocking rate, together with the selective hand chemical control of encroacher woody species, may be the most ecologically friendly and sustainable strategy to combat bush thickening and degradation and to improve the condition of the rangeland.

Kgosikoma *et al*. (2012) also found that high grass forage production and woody thickening were suppressed when implementing sustainable RGM methods. Other advantages of RGM include (a) reduced livestock losses due to illness, injury, or theft; (b) more tame animals, as they are monitored and handled more often; (c) flexibility in terms of taking advantage of unusual rainfall events due to the rotation of the animals to more productive areas and (d) the ability to rest areas for better fodder production in the dry season (Brunson & Burrit, 2009).

RGM in combination with HC can also ensure improved maintenance of the savanna structure since trees with varying heights and of differing species can be left uncontrolled, thereby ensuring their continued occurrence in the savanna ecosystem. In this way, a more stable environment and conditions favourable for the establishment of grass plants can be created (Smit & Rethman, 2000; Smit, 2004).

Large woody plants retained in HC sites are also able to suppress the establishment of new woody saplings, thereby reducing the woody-grass competition whilst retaining the beneficial effects of the woody plants in terms of soil enrichment and the provision of shade and food for browsing animals (Smit *et al.* 1996; Smit, 2004; Hagos & Smit, 2005). A significant portion of browse-able material and pods from leguminous woody plants that are normally part of the BT sites can also be utilised by cattle and game during the dry season, even when abundant grass is available (Kelly, 1977; Donaldson, 1979; Fagg & Stewart, 1994). Sound grazing management actions (i.e. RGM with long periods of rest), especially during the rainy season, ought to ensure a competitive grass layer to maintain a productive grazing rangeland system (Smit, 2004).

Since the re-thickening of trees at all bush-controlled sites will always be possible, a control programme for the management and mitigation of encroaching vegetation needs to be implemented when mitigating bush thickening (Barac, 2003; Campbell, 2000). It is further suggested that (a) an initial control be implemented to ensure the drastic reduction of the existing population (e.g. if large areas are thickened by woody plants, it may be more cost effective to remove the plants chemically with an aeroplane), (b) a follow-up control be implemented that will control newly established woody saplings and coppice re-growth (e.g. selective hand application of arboricides on targeted encroacher species only) and (c) maintenance control be implemented to maintain a low number of unwanted woody plant species with low annual control costs (e.g. only treating existing small problem areas by selective hand application of arboricides every four years and maintaining a high competitive grass layer with a sustainable RGM system). If these simple actions are applied, the thickening of unwanted woody plants would no longer be considered a problem.

The land user should realise the importance of implementing a good, long-term management system after chemical control actions have been applied. The recommendation is to prohibit any grazing during the first year and to only allow light grazing in the second and third year following on the control of woody species (Van der Merwe & Kellner, 1999). The amount of rainfall, the condition of the grass layer of rangelands surrounding cleared sites (i.e. nearby seed production and dispersal) and the recovery potential of the established species after the previous growing season are factors that ought to be considered when determining the extent of

grazing and stocking rates over the long term following the control of bush thickening (O'Connor & Roux, 1995; Fynn & O'Connor, 2000; Quaas *et al*., 2007).

From this study, it is evident that not all land users can afford or are able to implement RGM or chemical control actions to reduce high woody plant densities. Thus, it is proposed that when grazing becomes limited during the winter months or in below-average rainfall seasons, the forage production by the woody component can become a more important source of nutrition for both livestock and game, especially in the semi-arid savanna regions, such as the Molopo (Leggett *et al*., 2003). Different herbivore species including grazers (cattle and buffalo) and browsers (goats, kudu and giraffe), ruminants (cattle and sheep) and non-ruminants (horses and donkeys) with varying digestive systems can then be used in the management system (Smet & Ward, 2005). The feeding strategies and forage preferences of the respective animals are influenced by their digestive systems. Thus, by stocking an array of herbivore species, the ability of a management system to exploit different vegetation types occurring in a rangeland can be enhanced (Owen-Smith, 1999).

Herbivore diversity could also have a stabilising effect on rangeland productivity with the continuously changing vegetation, as often experienced in these non-equilibrium rangeland systems (Smet & Ward, 2005). Studies conducted by Smet and Ward (2005) on game farms with a high diversity of herbivore species showed less degradation around water-points (i.e. high abundances of perennial grass species and the lowest shrub densities). Land users may consider diversifying their type of livestock animals (i.e. browser and grazer animals), as large numbers of browser species can fulfil a vital role in the management and mitigation of bush thickening (Angassa, 2005; Smet & Ward, 2005). This land use can be considered for the BT areas that are not too dense during the development of a management strategy and when expensive chemical control actions cannot be applied.

If woody plants are cleared during bush control operations, it is recommended that a 30% woody cover be maintained on the rangeland (Richter, personal

communication[8]) as woody plants can become a very important food source during drought periods, create suitable habitats for grass establishment, favour the animals by providing shade and shelter and heighten the aesthetic appeal of the site.

The main reason why chemical control methods are not implemented is the limitation imposed by high costs. However, evaluating the total economic implications of the two chemical bush control applications (HC and AC) fell beyond the scope of this study. Nevertheless, the results of this study may provide essential quantitative data for a thorough economical evaluation which ought to be conducted in future.

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Appendix

Step 3 of the integrated assessment protocol being implemented with member from the multi-stakeholder platform of the Molopo study area (**Figure A1**).

Examples of graphical representation of (a) overall partial order (action number), where the relative outranking relationships between actions are epicted and (b) criteria (indicator) for which each action outranks the other actions (Source: Buatista & Orr, 2011) (**Figure A2**).

Evaluation of restoration and management actions on a Likert scale (Step 6). This step was conducted by members of the multi-stakeholder platform (**Figure A3**).

An example of a disc pasture meter (**Figure A4**).

Plastic poles were used for biometric measurements at each woody plant (**Figure A5**).

A manual soil auger was used to collect soil samples from the 30 cm top soil layer (**Figure A6**).

Illustrations of the different restoration and management sites (**Figures A7 – A12**).

Effect of rotational grazing, chemical hand control and chemical aeroplane control actions as compared to bush thickened sites on all assessed soil parameters (**Figure A13**).

Overview table of the 50 quantitative vegetation assessments sites with the corresponding restoration and management practices (**Table A1**).

Stakeholder categories from the multi-stakeholder platform (**Table A2**).

Grass species allocations to six grass functional group (GFG) (**Tables A3-A4**).

Effect of restoration and management actions as compared to bush thickened sites on the relative frequency distribution of woody plant species (**Table A5**).

Total of 41 identified species accounting for vegetation dissimilarity between the restoration and management sites as calculated by SIMPER (**Tables A6 – A11**).

Effect of rotational grazing management, chemical hand control and chemical aeroplane control actions as compared to bush thickened sites on the relative frequency distribution of grass functional groups (GFG's). (**Table A12**).

An example of the questionnaire used during step 1 and 2 of the integrated assessment protocol (**Data sheet A1**).

An example of the data sheet used during the two workshops (Step 3a and b of the IAPro) to record the ranking of indicators from each stakeholder (**Data sheet A2**).

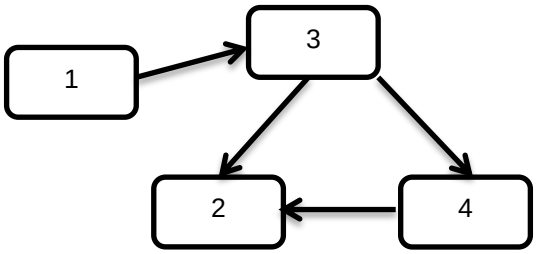
An example of the questionnaire used to gain information from the land user on the type of animals on the farm, stocking rate, the total time period camps were rested, type of chemical treatment, time between treatments and dosage used **(Data sheet A3).**

An example of the data sheet used during the two workshops for Step 6 of the IAPro to record qualitative data on the ranking of actions (**Data sheet A4**).

Harmse, C.J., Dreber, N., Kellner, K. & Kong, T.M. 2013. Linking farmer knowledge and biophysical data to evaluate actions for land degradation mitigation in savanna rangelands of the Molopo, South Africa. *In*: Revitalising grasslands to sustain our communities: Proceedings of the 22nd International Grassland Congress. Sydney, Australia, 15-19 September 2013. (Eds. D.L. Michalk, G.D. Millar, W.B. Badgery and Broadfoot, K.M.) pp. 1128-1131. (**Publication follows after appendix**).

Figure A1: Step 3 of the integrated assessment protocol being implemented with members from the multi-stakeholder platform.

a)



b)

	Ind A	Ind B	Ind C	Ind D	Ind E
Action 1	■	■	■	■	
Action 2					■
Action 3	■	■			
Action 4		■	■		

Figure A2: Examples of graphical representation of (a) overall partial order (action number), where the relative outranking relationships between actions are epicted and (b) criteria (indicator) for which each action outranks the other actions (Source: Buatista & Orr, 2011).

Figure A3: Evaluation of restoration and management actions on a Likert scale (Step 6). This step was conducted by members of the multi-stakeholder platform.

Figure A4: The disc pasture meter being used and explained to a land-user on how to determine the grass forage production.

Figure A5: Plastic poles that were 2 m long and marked in 10 cm intervals were used for biometric measurements at each woody plant.

Figure A6: A manual soil auger was used to collect soil samples from the 30 cm top soil layer.

a) b)

Figure A7: Illustration of the bush thickened sites. Photos were taken during March 2013. GPS coordinates (a) S -25.600; E 23.258, and (b) S -26.606; E 23.958

a) b)

Figure A8: Illustration of the aeroplane chemical controlled sites. Photo (a) was taken during February 2012 and (b) during March 2013. GPS coordinates (a) S -25.484; E 23.511, and (b) S -25.375; E 23.384

a) b)

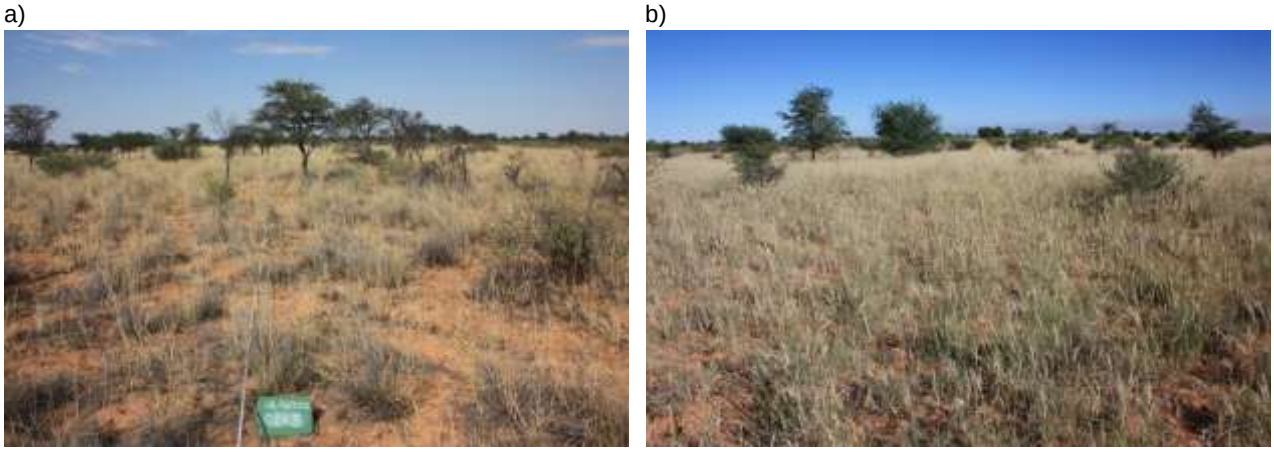


Figure A9: Illustration of the hand chemical controlled sites. Photo (a) was taken during March 2013 and photo (b) during January 2012. GPS coordinates (a) S -25.537; E 23.436, (b) S -26.549; E 22.568

a) b)

Figure A10: Illustration of the re-vegetated sites. Photos were taken during February 2012. GPS coordinates (a) S -26.159; E 23.906, and (b) S -26.567; E 22.578

a) b)

Figure A11: Illustration of the rotational grazing management sites. Photos were taken during February 2012. GPS coordinates (a) S -26.593; E 23.975, and (b) S -26.554; E 23.831

a) b)

Figure A12: Illustration of the continuous grazed sites. Photos were taken during March 2013. GPS coordinates (a) S -26.737; E 23.978, and (b) S -26.567; E 24.000

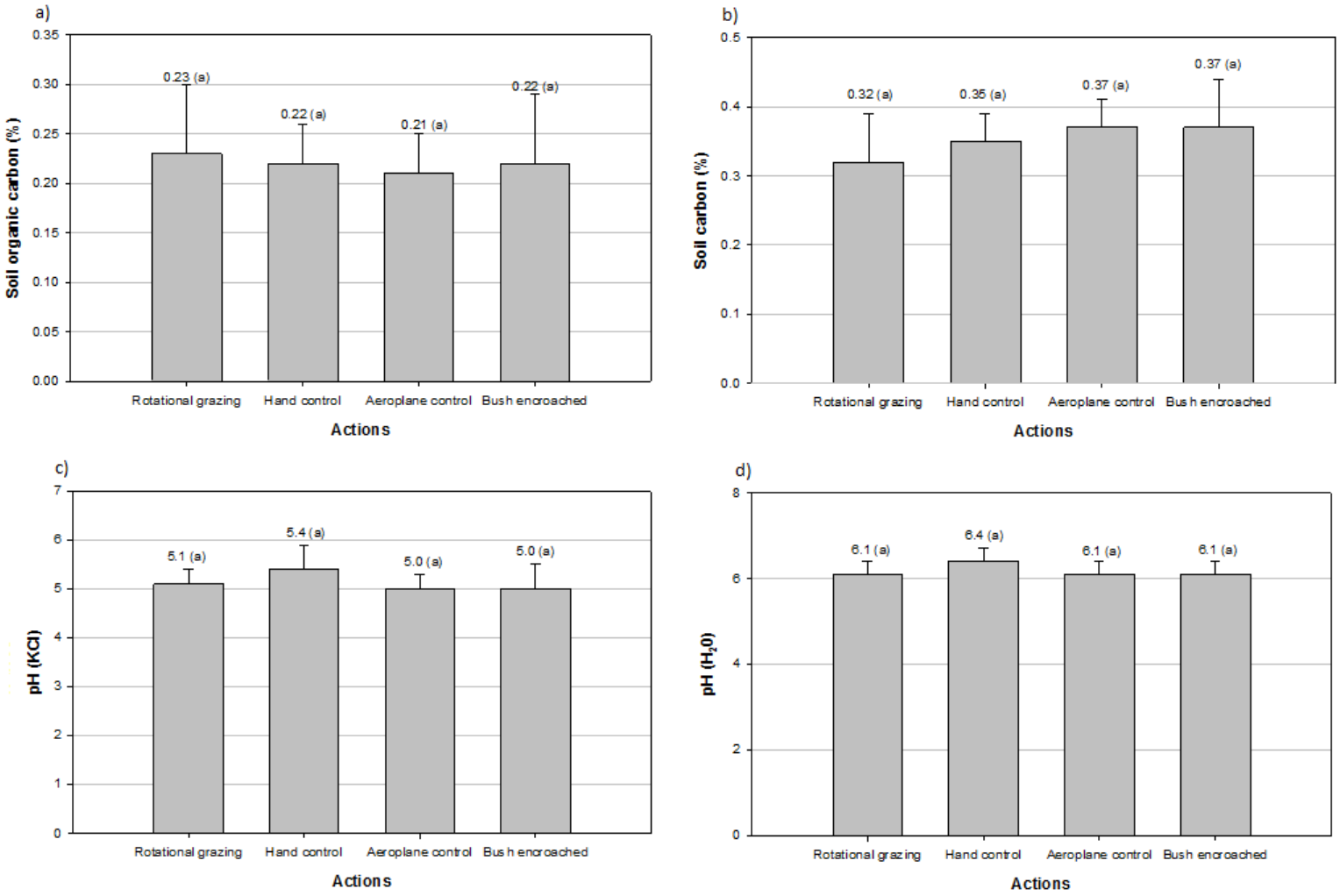


Figure A13: Effect of rotational grazing (n = 8), chemical hand control (n = 7) and chemical aeroplane control (n = 7) actions as compared to bush thickened sites (n = 15) on (a) the soil organic carbon (%), (b) soil carbon (%), (c) pH (KCl) and (d) pH (H_20). Bars indicate mean values with standard deviations. Different lower-case letters indicate a significant difference at $p < 0.05$ (ANOVA with post-hoc Tukey's HSD test).

Table A1: Overview table of the 50 quantitative vegetation assessments surveys with the corresponding restoration and management practices.

			Survey site (Coordinates)	
Stakeholder	**Stakeholder category**	**Restoration/Management type**	**S**	**E**
Mr. H. Killian	Research – Game Reserve Manager	Hand control 1	-25.585	23.342
		Hand control 2	-25.545	23.14
		Aeroplane control 1	-25.505	23.272
		Bush-thickened 1	-25.600	23.258
		Bush-thickened 2	-25.563	23.280
		Bush-thickened 3	-25.459	23.254
Mr. G. Keyser	Landowner- Commercial (Livestock)	Aeroplane control 1	-25.484	23.511
		Aeroplane control 2	-25.492	23.503
		Aeroplane control 3	-25.487	23.509
		Hand control 1	-25.536	23.460
		Hand control 2	-25.553	23.409
		Hand control 3	-25.537	23.436
		Bush-thickened 1	-25.489	23.496
		Bush-thickened 2	-25.47	23.512
		Bush-thickened 3	-25.483	23.501
Mr. E. Graupner	Landowner- Commercial (Game)	Bush-thickened 1	-25.321	23.367
		Bush-thickened 2	-25.323	23.388
		Bush-thickened 3	-25.332	23.401
		Aeroplane control 1	-25.384	23.380
		Aeroplane control 2	-25.375	23.384
		Aeroplane control 3	-25.362	23.375
Mr. D. Seralpelwane	Lease- Small commercial (Livestock)	Bush-thickened 1	-26.026	22.706
		Bush-thickened 2	-26.034	22.708
		Rotational grazing management1	-26.041	22.740
Mr. J. Tekolo	Communal- Tribal (Livestock)	Continuous grazing management 1	-26.737	23.978
		Continuous grazing management 2	-26.751	24.000
		Bush-thickened 1	-26.731	23.982
Mr. C. Nkokou	Lease- Subsistent (Livestock)	Rotational grazing management 1	-26.593	23.975
		Rotational grazing management 2	-26.617	23.959
		Bush-thickened 1	-26.606	23.958
		Bush-thickened 2	-26.606	23.958
Mr. T. Mongwaketsi	Lease- Subsistent (Livestock)	Rotational grazing management 1	-26.554	23.831
		Rotational grazing management 2	-26.561	23.821
		Bush-thickened 1	-26.544	23.862
Mr. J. Disipi	Communal- Tribal	Bush-thickened 1	-26.664	23.885
Mr. F. Setae	Communal- Tribal (Livestock)	Bush thickened 1	-26.676	23.882
		Continuous grazing management 1	-26.676	23.882
Mr. G. Olivier	Landowner- Commercial (Livestock)	Re-vegetation 1	-26.830	22.823
		Hand control 1	-26.843	22.861
Mr. L. Hauman	Landowner- Commercial (Livestock)	Rotational grazing management 1	-26.721	22.766
		Rotational grazing management 2	-26.689	22.750

Stakeholder	Stakeholder category	Restoration/Management type	Survey site (Coordinates) S	E
Mr. J. Olivier	Landowner- Commercial (Livestock)	Re-vegetation 1	-26.567	22.578
		Bush thickened 1	-26.549	22.549
		Hand control 1	-26.549	22.568
Mr. N. Muller	Landowner- Small Commercial (Livestock)	Re-vegetation 1	-26.159	23.906
Mr. P. Bruwer	Landowner- Commercial (Livestock)	Rotational grazing management 1	-25.812	22.820

Table A2: Stakeholder categories of the multi-stakeholder platform.

Stakeholder category	Definition
Semi-commercial lease tenure systems	A certain area of land is granted by the government to certain individuals who hold the rights to use some parts of the land under leasehold for commercial production. The land holders of lease tenure systems do not own the land but have exclusive rights to their holding. They may develop infrastructure (e.g. fences and watering points) and are allowed to exclude other people from their land which they manage. Although the land under semi-commercial lease tenure systems is smaller than 8,000 ha, they may produce and generate income greater than subsistence (i.e. additional income) from either farming operations, which include livestock keeping, hunting of wildlife and ecotourism.
Small-scale communal tenure systems	Most grazing land not under a leasehold system is used communally as an open access, small-scale communal tenure systems. Such land is granted to a community by government, mainly for subsistence livestock farming. Whole communities therefore hold the rights to use parts of the land for grazing of their livestock and the utilization of other natural resources, such as wood for fuel and construction. A land user in this system has a right to farm in a tribal communal farm as a resident or when accepted as a member of the tribe.
Small scale-LRAD farms	Small scale-LRAD farms are granted to small family groups and the farm. The land is owned collectively and managed through a committee, operating at a subsistence level.
Commercial-private systems	Commercial private land users own the land by obtaining cart and transport papers as issued by the Provincial Government. These land users normally farm at a large scale (>8,000 ha), including different livestock and game species.
Governmental experts	Agriculture extension officers, managers or researchers working for a provincial department in Agriculture and Rural Development.
Conservation manager	Someone who is involved in conservation practices as part of a profession, such as an employee of a nature reserve or conservation consultant group.
Consultants	Hold expertise in rangeland management and animal production.
Researcher	Worked for an academic institution in rangeland ecology.

Table A3: Grass species allocations to six grass functional group (GFG) types grouped according to their life cycle, palatability and response type from; van Oudtshoorn's (2012) guide to grass species classification for southern Africa, personal observations and in case of doubt the opinion of local land users regarding the palatability and plant vigour of a particular species was considered.

Life cycle	Perennial (1), weak-perennial (2) and annual plant (3).
Palatability	Palatable (1) and unpalatable (2) to livestock and game. Palatability can be defined as the plant material available and preferred by the herbivore animal influenced by digestibility and nutritional value of the plant species (Scholes, 2009; van Oudtshoorn, 2012). However, if herbivores had no choice then they will eat plant species (except if they are toxic) they would usually ignore under good veld conditions (Scholes, 2009). In southern Africa the digestibility range of grass plants is restricted between 40 – 60% (Scholes, 2009).
Response type	Many South African grass plant species has been defined according to a response curve to grazing pressure in various ecological status groups .e.g. Decreaser (4), Increaser 1, 2 and 3 type species (Hardy *et al.*, 1999; Scholes, 2009; van Oudtshoorn, 2012). The ecological status groups each predict which group of species will become dominant in areas where the grazing is light, medium or heavy and this is used in most of the South African veld condition assessments (Scholes, 2009).

Table A4: Grass functional group classification according to their life cycle, palatability and response type.

Grass species	Life cycle	Palatability	Response type	Grass functional group
Anthephora argentea	1	1	1	GFG1 - perennial, palatable climax grass, largely grazing sensitive
Anthephora pubescence	1	1	4	GFG1 - perennial, palatable climax grass, largely grazing sensitive
Brachiaria nigropedata	1	1	4	GFG1 - perennial, palatable climax grass, largely grazing sensitive
Centropodia glauca	1	1	4	GFG1 - perennial, palatable climax grass, largely grazing sensitive
Digitaria eriantha	1	1	4	GFG1 - perennial, palatable climax grass, largely grazing sensitive
Schmidtia pappophoroides	1	1	4	GFG1 - perennial, palatable climax grass, largely grazing sensitive
Eragrostis lehmanniana	1	1	2	GFG 2 - perennial, palatable pioneer-sub climax grass, grazing tolerant
Eragrostis nindensis	1	1	2	GFG 2 - perennial, palatable pioneer-sub climax grass, grazing tolerant
Stipagrostis uniplumis	1	1	2	GFG 2 - perennial, palatable pioneer-sub climax grass, grazing tolerant
Aristida meridionalis	1	2	3	GFG 3 - perennial, unpalatable climax grass, largely grazing tolerant
Eragrostis pallens	1	2	3	GFG 3 - perennial, unpalatable climax grass, largely grazing tolerant
Aristida stipitata	1	2	2	GFG 3 - perennial, unpalatable pioneer-sub climax grass, grazing tolerant
Stipagrostis hirtigluma	1	2	2	GFG 3 - perennial, unpalatable pioneer-sub climax grass, grazing tolerant
Triraphis schinzii	1	2	1	GFG 3 - perennial, unpalatable climax grass, largely grazing tolerant
Aristida congesta	2	2	2	GFG 4 - weak perennial, unpalatable pioneer sub-climax grass, grazing tolerant
Eragrostis trichophora	2	2	2	GFG 4 - weak perennial, unpalatable pioneer sub-climax grass, grazing tolerant
Urochloa mosambicensis	2	2	2	GFG 4 - weak perennial, unpalatable pioneer sub-climax grass, grazing tolerant
Melinis repens	2	2	2	GFG 5 - weak perennial, unpalatable pioneer sub-climax grass, grazing tolerant
Pogonarthria squarrosa	2	2	2	GFG 5 - weak perennial, unpalatable pioneer sub-climax grass, grazing tolerant
Schmidtia kalahariensis	3	2	2	GFG 6 - annual, unpalatable pioneer sub-climax grass, grazing tolerant
Tragus berteronianus	3	2	2	GFG 6 - annual, unpalatable pioneer sub-climax grass, grazing tolerant

Table A5: Effect of RGM (n = 8), chemical HC (n = 7) and chemical AC (n = 7) actions as compared to BE sites (n = 15) on the relative frequency distribution (%) of woody plant species. Means (±SD) with different lower-case letters in a row indicate a significant difference at $p < 0.001$ (Kruskal-Wallis test with Mann-Whitney post-hoc pairwise test).

	Rotational grazing management (RGM)	**Hand chemical control (HC)**	**Aeroplane chemical control (AC)**	**Bush thickened (BT)**
Acacia erioloba	40.8±33.1[a]	28.1±18.6[a]	10.1±13.5[a,b]	2.5±3.2[b]
Acacia haematoxylon	4.4±6.7[a]	1.8±4.1[a]	0.0±0.0	0.1±0.4[a]
Acacia hebeclada	2.9±6.0	0.9±1.2[a]	0.0±0.0	0.3±0.9[a]
Acacia luederitzii	5.0±5.6[a]	0.0±0.0	5.3±5.2[a]	5.9±7.2[a]
Acacia mellifera	12.7±13.8[a]	17.0±14.1[a]	15.1±17.0[a]	38.2±22.9[a]
Boscia albitrunca	12.7±13.8[a]	3.6±4.8[a]	37.7±26.6[b]	3.8±3.2[a]
Dichrostachys cinerea	10.5±18.8[a]	0.8±1.4[a]	1.9±2.7 [a]	10.8±19.6 [a]
Diospyros lycioides	0.0±0.0	0.6±1.0±[a]	3.6±5.9±[a]	0.5±1.6[a]
Grewia flava	16.8±20.8[a]	33.6±17.8[a]	17.3±17.3[a]	21.7±11.1[a]
Grewia retinervis	0.0±0.0	0.0±0.0	1.1±2.1[a]	0.6±1.4[a]
Gymnosporia buxifolia	0.3±0.9[a]	0.0±0.0	0.0±0.0	1.8±5.2[a]
Lycium cinereum	0.3±0.7[a]	1.8±2.3[a]	2.6±4.0[a]	2.0±3.5[a]
Rhigozum brevispinosum	0.0±0.0	7.7±16.0[a]	2.5±3.8[a]	2.1±5.7[a]
Searsia tenuinervis	0.7±1.9[a]	1.1±1.1[a]	2.1±3.4[a]	0.7±1.7[a]
Tarchonanthus camphoratus	1.2±2.5[a]	0.0±0.0	0.0±0.0	2.1±6.5[a]
Terminalia sericea	0.7±1.4[a]	0.0±0.0	0.0±0.0	5.3±10.9[a]
Ziziphus mucronata	0.2±0.6[a]	3.0±3.6[a]	0.6±1.0[a]	0.8±1.3[a]

Table A6: The total of 41 identified plant species accounting for vegetation dissimilarity between aeroplane control (AC) and bush thickened (BT) sites as calculated by SIMPER. 'Contribution %' indicates the average percentage contribution from the ith species to the overall Bray–Curtis-dissimilarity between BE and AC sites. "Cumulative %" indicates the cumulative percentage contribution to the total dissimilarity.

Species	Mean abundance		Contribution (%)	Cumulative (%)
	BE	AC		
Acacia mellifera	47.1	15.1	13.6	13.6
Boscia albitrunca	5.2	37.7	13.4	27.0
Schmidtia kalahariensis	28.9	0.1	11.8	38.8
Schmidtia pappophoroides	0.5	20.7	8.3	47.1
Eragrostis lehmanniana	18.1	27.6	7.7	54.8
Grewia flava	23.0	17.3	7.0	61.9
Urochloa mosambicensis	14.9	13.7	6.9	68.8
Stipagrostis uniplumis	16.6	17.9	4.6	73.4
Aristida stipitata	10.7	3.3	4.1	77.5
Acacia erioloba	3.6	10.1	3.8	81.2
Melinis repens	6.3	9.8	3.0	84.2
Acacia luederitzii	7.6	5.3	2.7	86.9
Rhigozum brevispinosum	2.7	2.5	1.7	88.6
Diospyros lycioides	0.7	3.6	1.6	90.1
Lycium cinereum	2.1	2.6	1.4	91.6
Eragrostis trichophora	1.7	2.1	1.1	92.7
Searsia tenuinervis	1.0	2.1	1.0	93.7
Terminalia sericea	2.4	0.0	1.0	94.7
Dichrostachys cinerea	1.5	1.9	1.0	95.6
Brachiaria nigropedata	0.0	1.7	0.7	96.3
Grewia retinervis	0.9	1.1	0.6	97.0
Ziziphus mucronata	1.1	0.6	0.5	97.5
Triraphis schinzii	1.2	0.0	0.5	98.0
Aristida congesta	0.3	0.9	0.4	98.4
Pogonarthria squarrosa	0.0	0.9	0.4	98.8
Aristida meridionalis	0.6	0.3	0.3	99.1
Megaloprotachne albescence	0.0	0.4	0.2	99.3
Ehretia rigida	0.3	0.0	0.1	99.4
Acacia hebeclada	0.3	0.0	0.1	99.6
Tarchonanthus camphoratus	0.3	0.0	0.1	99.7
Tragus berteronianus	0.0	0.3	0.1	99.8
Gymnosporia buxifolia	0.2	0.0	0.1	99.9
Cadaba aphylla	0.1	0.0	0.1	99.9
Digitaria eriantha	0.0	0.1	0.1	100.0
Centropodia glauca	0.0	0.0	0.0	100.0
Acacia karroo	0.0	0.0	0.0	100.0
Acacia haematoxylon	0.0	0.0	0.0	100.0
Eragrostis pallens	0.0	0.0	0.0	100.0
Stipagrostis hirtigluma	0.0	0.0	0.0	100.0
Anthephora pubenscens	0.0	0.0	0.0	100.0
Anthephora argentea	0.0	0.0	0.0	100.0

Table A7: The total of 41 identified species accounting for vegetation dissimilarity between rotational grazing management (RGM) and bush thickened (BT) sites as calculated by SIMPER. 'Contribution %' indicates the average percentage contribution from the ith species to the overall Bray–Curtis-dissimilarity between BE and RGM sites. "Cumulative %" indicates the cumulative percentage contribution to the total.

Species	Mean abundance		Contribution (%)	Cumulative (%)
	BE	RGM		
Schmidtia pappophoroides	0.5	56.7	18.9	18.9
Acacia erioloba	3.6	40.8	12.8	31.6
Acacia mellifera	47.1	12.7	11.8	43.4
Schmidtia kalahariensis	28.9	0.0	9.7	53.1
Grewia flava	23.0	16.8	6.7	59.8
Urochloa mosambicensis	14.9	0.1	5.0	64.8
Stipagrostis uniplumis	16.6	4.9	4.7	69.5
Eragrostis lehmanniana	18.1	14.5	4.1	73.5
Aristida stipitata	10.7	11.9	3.7	77.2
Dichrostachys cinerea	1.5	10.4	3.5	80.7
Acacia luederitzii	7.6	5.0	2.3	83.0
Melinis repens	6.3	0.3	2.1	85.1
Boscia albitrunca	5.2	0.8	1.6	86.7
Eragrostis pallens	0.0	4.6	1.6	88.3
Acacia haematoxylon	0.0	4.4	1.5	89.7
Acacia hebeclada	0.3	2.9	1.0	90.7
Acacia karroo	0.0	2.9	1.0	91.7
Terminalia sericea	2.4	0.7	0.9	92.7
Rhigozum brevispinosum	2.7	0.0	0.9	93.6
Lycium cinereum	2.1	0.3	0.7	94.3
Eragrostis trichophora	1.7	0.4	0.6	94.9
Anthephora pubescence	0.0	1.8	0.6	95.5
Searsia tenuinervis	1.0	0.7	0.5	96.0
Tarchonanthus camphoratus	0.3	1.2	0.5	96.5
Centropodia glauca	0.0	1.4	0.5	96.9
Triraphis schinzii	1.2	0.1	0.4	97.4
Ziziphus mucronata	1.1	0.2	0.4	97.8
Brachiaria nigropedata	0.0	1.1	0.4	98.1
Anthephora argentea	0.0	0.9	0.3	98.4
Grewia retinervis	0.9	0.0	0.3	98.7
Diospyros lycioides	0.7	0.0	0.2	99.0
Aristida meridionalis	0.6	0.1	0.2	99.2
Stipagrostis hirtigluma	0.0	0.6	0.2	99.4
Gymnosporia buxifolia	0.2	0.3	0.2	99.6
Aristida congesta	0.3	0.1	0.1	99.7
Ehretia rigida	0.3	0.0	0.1	99.8
Tragus berteronianus	0.0	0.3	0.1	99.9
Cadaba aphylla	0.1	0.0	0.0	100.0
Pogonarthria squarrosa	0.0	0.1	0.0	100.0
Megaloprotachne albescence	0.0	0.0	0.0	100.0
Digitaria eriantha	0.0	0.0	0.0	100.0

Table A8: The total of 41 identified species accounting for vegetation dissimilarity between bush thickened (BT) and hand control (HC) sites as calculated by SIMPER. 'Contribution %' indicates the average percentage contribution from the ith species to the overall Bray–Curtis-dissimilarity between BE and HC sites. "Cumulative %" indicates the cumulative percentage contribution to the total dissimilarity.

Species	Mean abundance		Contribution (%)	Cumulative (%)
	BE	HC		
Schmidtia pappophoroides	0.5	59.1	22.5	22.5
Acacia mellifera	47.1	17.0	11.9	34.3
Schmidtia kalahariensis	28.9	0.7	10.8	45.2
Acacia erioloba	3.6	28.1	9.5	54.7
Grewia flava	23.0	33.6	6.6	61.3
Urochloa mosambicensis	14.9	14.9	6.1	67.3
Eragrostis lehmanniana	18.1	6.4	5.4	72.8
Stipagrostis uniplumis	16.6	9.8	4.3	77.1
Aristida stipitata	10.7	4.4	4.1	81.2
Rhigozum brevispinosum	2.7	7.7	3.5	84.6
Acacia luederitzii	7.6	0.0	2.9	87.5
Melinis repens	6.3	3.2	2.1	89.7
Boscia albitrunca	5.2	3.6	1.8	91.4
Ziziphus mucronata	1.1	3.0	1.1	92.6
Lycium cinereum	2.1	1.8	1.1	93.7
Terminalia sericea	2.4	0.0	0.9	94.6
Eragrostis trichophora	1.7	0.4	0.7	95.3
Acacia haematoxylon	0.0	1.8	0.7	96.0
Dichrostachys cinerea	1.5	0.8	0.7	96.7
Searsia tenuinervis	1.0	1.1	0.6	97.2
Triraphis schinzii	1.2	0.0	0.5	97.7
Diospyros lycioides	0.7	0.6	0.4	98.1
Acacia hebeclada	0.3	0.9	0.4	98.5
Grewia retinervis	0.9	0.0	0.3	98.8
Aristida meridionalis	0.6	0.1	0.3	99.1
Brachiaria nigropedata	0.0	0.6	0.2	99.3
Aristida congesta	0.3	0.1	0.2	99.5
Ehretia rigida	0.3	0.0	0.1	99.6
Tarchonanthus camphoratus	0.3	0.0	0.1	99.8
Centropodia glauca	0.0	0.3	0.1	99.9
Gymnosporia buxifolia	0.2	0.0	0.1	99.9
Cadaba aphylla	0.1	0.0	0.1	100.0
Pogonarthria squarrosa	0.0	0.0	0.0	100.0
Acacia karroo	0.0	0.0	0.0	100.0
Megaloprotachne albescence	0.0	0.0	0.0	100.0
Eragrostis pallens	0.0	0.0	0.0	100.0
Tragus berteronianus	0.0	0.0	0.0	100.0
Stipagrostis hirtigluma	0.0	0.0	0.0	100.0
Digitaria eriantha	0.0	0.0	0.0	100.0
Anthephora pubescence	0.0	0.0	0.0	100.0
Anthephora argentea	0.0	0.0	0.0	100.0

Table A9: The total of 41 identified plant species accounting for the vegetation dissimilarity between rotational grazing management (RGM) and hand control (HC) sites as calculated by SIMPER. 'Contribution %' indicates the average percentage contribution from the ith species to the overall Bray–Curtis-dissimilarity between RGM and HC sites. "Cumulative %" indicates the cumulative percentage contribution to the total dissimilarity.

Species	Mean abundance		Contribution (%)	Cumulative (%)
	HC	RGM		
Acacia erioloba	28.1	40.8	14.9	14.9
Schmidtia pappophoroides	59.1	56.7	13.1	28.0
Grewia flava	33.6	16.8	12.3	40.3
Acacia mellifera	17.0	12.7	7.3	47.6
Urochloa mosambicensis	14.9	0.1	7.2	54.8
Dichrostachys cinerea	0.8	10.4	5.1	59.9
Eragrostis lehmanniana	6.4	14.5	5.0	64.9
Aristida stipitata	4.4	11.9	4.9	69.8
Stipagrostis uniplumis	9.8	4.9	4.5	74.3
Rhigozum brevispinosum	7.7	0.0	3.7	78.0
Acacia haematoxylon	1.8	4.4	2.4	80.5
Melinis repens	0.0	5.0	2.4	82.9
Boscia albitrunca	0.0	4.6	2.2	85.1
Acacia erioloba	3.6	0.8	1.8	86.9
Acacia hebeclada	0.9	2.9	1.5	88.5
Melinis repens	3.2	0.3	1.5	90.0
Acacia karroo	0.0	2.9	1.4	91.4
Ziziphus mucronata	3.0	0.2	1.4	92.8
Lycium cinereum	1.8	0.3	0.9	93.7
Anthephora pubescence	0.0	1.8	0.9	94.5
Brachiaria nigropedata	0.6	1.1	0.8	95.3
Searsia tenuinervis	1.1	0.7	0.7	96.0
Centropodia glauca	0.3	1.4	0.7	96.7
Tarchonanthus camphoratus	0.0	1.2	0.6	97.3
Anthephora argentea	0.0	0.9	0.4	97.8
Schmidtia kalahariensis	0.7	0.0	0.3	98.1
Terminalia sericea	0.0	0.7	0.3	98.5
Eragrostis trichophora	0.4	0.4	0.3	98.8
Stipagrostis hirtigluma	0.0	0.6	0.3	99.1
Diospyros lycioides	0.6	0.0	0.3	99.4
Gymnosporia buxifolia	0.0	0.3	0.2	99.5
Tragus berteronianus	0.0	0.3	0.1	99.7
Aristida meridionalis	0.1	0.1	0.1	99.8
Aristida congesta	0.1	0.1	0.1	99.9
Pogonarthria squarrosa	0.0	0.1	0.1	99.9
Triraphis schinzii	0.0	0.1	0.1	100.0
Cadaba aphylla	0.0	0.0	0.0	100.0
Megaloprotachne albescence	0.0	0.0	0.0	100.0
Grewia retinervis	0.0	0.0	0.0	100.0
Ehretia rigida	0.0	0.0	0.0	100.0
Digitaria eriantha	0.0	0.0	0.0	100.0

Table A10: The total of 41 identified plant species accounting for the vegetation dissimilarity between aeroplane control (AC) and hand control (HC) sites as calculated by SIMPER. 'Contribution %' indicates the average percentage contribution from the ith species to the overall Bray–Curtis-dissimilarity between AC and HC. "Cumulative %" indicates the cumulative percentage contribution to the total dissimilarity.

Species	Mean abundance		Contribution (%)	Cumulative (%)
	AC	HC		
Schmidtia pappophoroides	20.7	59.1	17.6	17.6
Boscia albitrunca	37.7	3.6	14.3	31.9
Grewia flava	17.3	33.6	9.6	41.5
Acacia erioloba	10.1	28.1	9.4	50.9
Eragrostis lehmanniana	27.6	6.4	9.1	59.9
Acacia mellifera	15.1	17.0	6.9	66.8
Urochloa mosambicensis	13.7	14.9	6.1	72.9
Stipagrostis uniplumis	17.9	9.8	4.8	77.8
Rhigozum brevispinosum	2.5	7.7	3.6	81.3
Melinis repens	9.8	3.2	3.3	84.7
Aristida stipitata	3.3	4.4	2.2	86.9
Acacia luederitzii	5.3	0.0	2.2	89.1
Diospyros lycioides	3.6	0.6	1.5	90.6
Lycium cinereum	2.6	1.8	1.3	91.9
Ziziphus mucronata	0.6	3.0	1.2	93.1
Searsia tenuinervis	2.1	1.1	1.0	94.1
Eragrostis trichophora	2.1	0.4	0.9	95.0
Brachiaria nigropedata	1.7	0.6	0.9	95.8
Dichrostachys cinerea	1.9	0.8	0.9	96.7
Acacia haematoxylon	0.0	1.8	0.8	97.5
Grewia retinervis	1.1	0.0	0.5	97.9
Aristida congesta	0.9	0.1	0.4	98.3
Acacia hebeclada	0.0	0.9	0.4	98.7
Pogonarthria squarrosa	0.9	0.0	0.4	99.0
Schmidtia kalahariensis	0.1	0.7	0.3	99.4
Megaloprotachne albescence	0.4	0.0	0.2	99.5
Aristida meridionalis	0.3	0.1	0.2	99.7
Tragus berteronianus	0.3	0.0	0.1	99.8
Centropodia glauca	0.0	0.3	0.1	99.9
Digitaria eriantha	0.1	0.0	0.1	100.0
Cadaba aphylla	0.0	0.0	0.0	100.0
Acacia karroo	0.0	0.0	0.0	100.0
Terminalia sericea	0.0	0.0	0.0	100.0
Triraphis schinzii	0.0	0.0	0.0	100.0
Eragrostis pallens	0.0	0.0	0.0	100.0
Tarchonanthus camphoratus	0.0	0.0	0.0	100.0
Gymnosporia buxifolia	0.0	0.0	0.0	100.0
Stipagrostis hirtigluma	0.0	0.0	0.0	100.0
Ehretia rigida	0.0	0.0	0.0	100.0
Anthephora pubescence	0.0	0.0	0.0	100.0
Anthephora argentea	0.0	0.0	0.0	100.0

Table A11: The total of 41 identified plant species accounting for the vegetation dissimilarity between aeroplane control (AC) and rotational grazing management (RGM) sites as calculated by SIMPER. 'Contribution %' indicates the average percentage contribution from the ith species to the overall Bray–Curtis-dissimilarity between AC and RGM. "Cumulative %" indicates the cumulative percentage contribution to the total dissimilarity.

Species	Mean abundance		Contribution (%)	Cumulative (%)
	AC	RGM		
Schmidtia pappophoroides	15.0	56.7	20.7	15.0
Boscia albitrunca	13.6	0.8	37.7	28.6
Acacia erioloba	13.1	40.8	10.1	41.7
Grewia flava	7.2	16.8	17.3	48.9
Eragrostis lehmanniana	6.0	14.5	27.6	54.9
Acacia mellifera	5.8	12.7	15.1	60.7
Stipagrostis uniplumis	5.6	4.9	17.9	66.3
Urochloa mosambicensis	5.0	0.1	13.7	71.3
Dichrostachys cinerea	3.8	10.4	1.9	75.1
Melinis repens	3.5	0.3	9.8	78.7
Aristida stipitata	3.5	11.9	3.3	82.2
Acacia luederitzii	2.1	5.0	5.3	84.3
Eragrostis pallens	1.7	4.6	0.0	86.0
Acacia haematoxylon	1.6	4.4	0.0	87.6
Diospyros lycioides	1.3	0.0	3.6	88.9
Acacia karroo	1.1	2.9	0.0	90.0
Acacia hebeclada	1.0	2.9	0.0	91.1
Lycium cinereum	1.0	0.3	2.6	92.0
Brachiaria nigropedata	0.9	1.1	1.7	93.0
Rhigozum brevispinosum	0.9	0.0	2.5	93.9
Searsia tenuinervis	0.9	0.7	2.1	94.7
Eragrostis trichophora	0.8	0.4	2.1	95.5
Anthephora pubescence	0.6	1.8	0.0	96.2
Centropodia glauca	0.5	1.4	0.0	96.7
Tarchonanthus camphoratus	0.4	1.2	0.0	97.1
Grewia retinervis	0.4	0.0	1.1	97.5
Aristida congesta	0.3	0.1	0.9	97.9
Anthephora argentea	0.3	0.9	0.0	98.2
Pogonarthria squarrosa	0.3	0.1	0.9	98.5
Terminalia sericea	0.3	0.7	0.0	98.8
Ziziphus mucronata	0.3	0.2	0.6	99.0
Stipagrostis hirtigluma	0.2	0.6	0.0	99.3
Tragus berteronianus	0.2	0.3	0.3	99.4
Megaloprotachne albescence	0.2	0.0	0.4	99.6
Aristida meridionalis	0.1	0.1	0.3	99.7
Gymnosporia buxifolia	0.1	0.3	0.0	99.9
Schmidtia kalahariensis	0.1	0.0	0.1	99.9
Digitaria eriantha	0.1	0.0	0.1	100.0
Triraphis schinzii	0.0	0.1	0.0	100.0
Cadaba aphylla	0.0	0.0	0.0	100.0
Ehretia rigida	0.0	0.0	0.0	100.0

Table A12: Effect of rotational grazing management (n = 8), chemical hand control (n = 7) and chemical aeroplane control (n = 7) actions as compared to bush thickened sites (n = 15) on the relative frequency distribution of grass functional groups (GFG's). Different lower-case letters in a row indicate a significant difference at $p < 0.001$ (Kruskal-Wallis test with Mann-Whitney post-hoc pairwise test).

Grass functional group	Rotational grazing management	Hand chemical control	Aeroplane chemical control	Bush thickened
1. perennial, palatable climax grass, largely grazing sensitive	61.9±16.5[a]	60.0±27.2[a]	19.1±21.1[b]	0.5±1.5[c]
2. perennial, palatable pioneer-subclimax grass, grazing tolerant	19.4±14.1[a]	16.1±12.4[a]	45.7±15.7[b]	34.7±2.6[b]
3. perennial, unpalatable climax grass, largely grazing tolerant	17.4±12.7[a]	4.5±7.1[b]	3.6±3.4[b]	12.5±11.5[a,b]
4. weak perennial, palatable pioneer-subclimax grass, grazing tolerant	0.6±1.1[a]	15.5±13.3[b]	16.8±13.7[b]	17.1±18.1[b]
5. weak perennial, unpalatable pioneer-subclimax grass, grazing tolerant	0.4±0.7[a]	3.2±3.9[a,b]	10.8±6.9[c]	6.3±5.6[b,c]
6. annual, unpalatable pioneer-subclimax grass, grazing tolerant	0.3±0.7[a]	0.7±1.3[a]	4.0±9.3[a]	28.9±9.9[b]

Data sheet A1: Questionnaire used during step 1 and 2 of the integrated assessment protocol (compiled by Dr Taryn Kong).

ID nr. .....

PRACTICE-Phase 1 and 2 Interview Discussion Note

1. ***Position***

Occupation/means to support living costs: *(if applicable)*

Title: *(if applicable)*

Organization: *(employer or write self-employed)*

Responsibilities: *(prompt for nature of responsibilities or resource usage and management at the study site. For farmers, can ask about how many farms they own/lease and how they utilize the farms.)*

Length and nature of experience: *(try to establish his/her level of expertise. For example, ask "How long do you stay on the dunes, permanently or only part of the time?" to the farmers in Kalahari.)*

Land right: *(refers to ownership)* **private / private-lease / public-lease / public**
Land regime: *(refers to management regime)* **individual / communal / open / not applicable**
Are you a member of any associations or groups in this region/site? Yes / No
If yes, what is the name of the group? Mission or objectives?

Others:

2. ***Ask the participant for a general description of the socio-environmental situation and conditions of the region or study site.***

3. ***Actions*** *(the aim is to elicit information concerning extent of knowledge about restoration/mitigation actions. Use the PRACTICE_Phases1_2_Discussion_Note_Tool.xls.*
Definition (list) of the actions to be evaluated *(At this stage, the information about the actions should be only about*

(1) the type of action

(2) the implementation time (when has it been applied? or is being applied?)

(3) the actual area where the actions were or are being applied. *(The list of actions should always include a "no action" option, where no mitigation/restoration action has been applied).*

If Yes: capture what is said.
What is the nature of your relationship or involvement?
Are these actions ongoing? If abandoned, why?

If No: the interviewer should inform the participant about the above actions and then try to find out to which extent the participant may be directly/indirectly related to the actions. For example, ask "From the description we have given, do you think these actions(s) can indirectly be related to you or can affect you?

4. Establishing stakeholder knowledge on the actions

5. Establishing the baseline personal evaluation on the

- *Do you know about the specifics of any of the listed actions as applied to this area, or do you know them in general sense?*
- *How would you rate your knowledge on each of the actions on a scale of 0 to 5, where 0 is "I don't know about this action" and 5 is "you know a lot"?* [9]
- *What is your main source of information about each of these actions?*

- *What do you think were the original goals of this action? What made you decide to use action XXX initially (if any was used by the participant)?*
- *In case this action is implemented again in the future, are there any other objectives that should be considered?*
- *How would you rate this action on a scale of 1 to 5, where 1 is "a very bad choice" and 5 is "an excellent choice," (use the table below to fill in their rating for individual action)?*
- *What do you feel are the positive aspects/outcomes?*[10]
- *What do you feel are the negative aspects/outcomes?*
- *If this action is implemented again in the future, how do you think it can be improved or would you use other actions?*
- *What's your general opinion on this action?*

6. Identifying indicators based on local-knowledge

7. Participation

Would you consider participating in the assessment of restoration/mitigation actions? Yes / No

[9] This rating of the stakeholders' knowledge applies to both types of knowledge: general (about the type of action) or site-specific (about the actual implementation of the action in the target site), as both can vary from not at all knowledgeable to extremely knowledgeable.

[10] The interviewer will take note of the positive and negative outcomes pointed out by the participants as potential indicators, which will be confirmed through the following section (Identifying indicators based on local-knowledge).

Why or why not? *(to elicit motivations and expectations for participation)*
If no, what might encourage you to participate? *(to find out the barriers to participation)*
If yes, how often would you be willing to meet over the next 12 months?
What locations would be best for a meeting?
Would you be willing to answer some questions by email or using an online form? Yes / No

8. Referrals *(to identify other potential stakeholders or special interest groups)*

Who do you recommend that we speak to next to obtain further information? *(Explain that we are looking for people who may know a lot about, been involved with, or been affected by the local environmental conditions and/or the restoration/mitigation actions.)*
Name *(if known)* & Affiliation/Network
Contact Information *(if known)*
Why do you recommend them? *(elicit knowledge, experience, and/or potential to represent a group or interest)*
Comments on prior experience with this person or organization
Comments about collaborative potential
Any concerns?

Please place any other names on the back of this page.

9. Past records *(This is to elicit participants' knowledge of prior research/studies done on the two sites that can provide relevant background information)*

10. General Demographics of Respondent *(to elicit basic biographical information)*

Age:	Gender: M / F	Preferred language *(add ethnic group if relevant)*:
Highest level of education completed: *(Ask with sensitivity, e.g., Do you have any schooling? Let the participant to volunteer his/her level of education.)* □ primary □ secondary □ professional training □ university □ beyond		

Note: This final page will only be used to for making contact with the potential stakeholder. The unique ID number on the top of this page must be added to all associated pages. In order to safeguard privacy and confidentiality, when the discussion is completed, this page will be stored separately from all other information collected associated with this individual. Once the need for this is past, that is, after the last stakeholder engagement activity, these contact information pages should be destroyed.

Interviewer Name:

Assistant Interviewer Name: Interpreter Name:

Study site name:

Potential Stakeholder Contact information

Participant Name:	Date
Phone No.:	Email address:
City/town/community of residence:	Address: home / work *(describe generally if no specific address)*
What is the best way to communicate with you? phone / email / regular mail / other:	

Closing Remarks

We thank you very much for sharing your valuable information with us. Should we need clarification on any of this information, can we contact you again? If you have any questions about our project, you can contact us at the numbers or email addresses printed on the pamphlet that we gave to you earlier. If you think of an additional person whom we should talk to, please contact us. Thank you!

Data sheet A2: The data sheet used during the two workshops (Step 3a and b of the IAPro) to record the ranking of indicators from each stakeholder.

Participant name:.....

SUBSET	N° of cards	Code of cards
1		
Blank cards between=		
2		
Blank cards between=		
3		
Blank cards between=		
4		
Blank cards between=		
5		
Blank cards between=		
6		
Blank cards between=		
7		
Blank cards between=		
8		
Blank cards between=		
9		
Blank cards between=		
10		
Blank cards between=		
11		
Blank cards between=		
12		

Z Ratio =

Datasheet A3: An example of the questionnaire used to gain information from the land user on the type of animals on the farm, stocking rate, the total time period camps were rested, type of chemical treatment, time between treatments and dosage used.

Farmer: _____ Date: _____

Type of livestock on farm?

How many animals, stocking rate?

Farm and camp size?

Rainfall data?

Fire frequency?

Farm history, e.g. Old fields

How long is veld or camp rested?

Chemical control

Frequency, quantities, chemical used, when was it treated

Datasheet A4: An example of the data sheet used during the two workshops for Step 6 of the IAPro to record qualitative data on the ranking of actions.

PRACTICE (Prevention and Restoration Actions to Combat Desertification: An Integrated Approach

Rangskikking van die verskillende restourasie en bestuurs en praktyke/ Ranking of different restoration and management practices

Molopo Werkswinkel, Kgokgojane, Noord Wes Provinsie – 5 Junie 2013

Molopo Workshop, Kgokgojane, North West Province – 5 June 2013

Naam/Name: _____.

NORTH-WEST UNIVERSITY
YUNIBESITI YA BOKONE-BOPHIRIMA
NOORDWES-UNIVERSITEIT
POTCHEFSTROOM CAMPUS

Kostes van materiaal en werk

Ditshenyegelo (di diriswa le badiri)

Costs (materials and labors)

Rank the 5 management actions, according to the impact each of the actions have on the cost for material and labor. (1 has the most positive impact, to 5 that has no or only a slight positive impact on the indicator).

Which action will have the most costs (1) regarding materials and labour to implement to the least costs (5)?

MANAGEMENT ACTION	**RANK (1 to 5)**
ROTATIONAL GRAZING	
REVEGETATION	
CHEMICAL BUSH CONTROL	
NO ROTATIONAL GRAZING	
BUSH ENCROACHED (NO BUSH CONTROL)	

Risiko's: droogtes of veld brande

Kotsi: molelo, go palelwa ke go tlhoga

Risks: drought or wildfires

Rank the 5 management actions, according to the impact each of the actions have on risks, e.g. droughts. (1 has the most positive impact, to 5 that has no or only a slight positive impact on the indicator).

e.g with which action will a farmer be best (1) prepared for an upcoming drought?

MANAGEMENT ACTION	RANK (1 to 5)
ROTATIONAL GRAZING	
REVEGETATION	
CHEMICAL BUSH CONTROL	
NO ROTATIONAL GRAZING	
BUSH ENCROACHED (NO BUSH CONTROL)	

Effektiwiteit van bos beheer: % doding en hergroei

Taolo ya ditlhare: go bolawa le go tlhoga gape

Bush control effectiveness: killing % and regrowth

Rank the 5 management actions, according to the impact each of the actions have on bush control effectiveness. (1 has the most positive impact, to 5 that has no or only a slight positive impact on the indicator).

Which action will be the most effective (1) to least effective (5) to prevent bush encroachment (1)?

MANAGEMENT ACTION	RANK (1 to 5)
ROTATIONAL GRAZING	
REVEGETATION	
CHEMICAL BUSH CONTROL	
NO ROTATIONAL GRAZING	
BUSH ENCROACHED (NO BUSH CONTROL)	

Inkomste en wins
Letseno le morokotso
Income and profit

Rank the 5 management actions, according to the impact each of the actions have on the income and profit the land user make (1 has the most positive impact, to 5 that has no or only a slight positive impact on the indicator).

Which action will ensure the highest (1) to lowest (5) income and profit after being implemented?

MANAGEMENT ACTION	RANK (1 to 5)
ROTATIONAL GRAZING	
REVEGETATION	
CHEMICAL BUSH CONTROL	
NO ROTATIONAL GRAZING	
BUSH ENCROACHED (NO BUSH CONTROL)	

Vee se kondisie en waarde
Maemo a pholofolo le boleng
Animal's condition and value

Rank the 5 management actions, according to the impact each of the actions have on the animal's condition and value. (1 has the most positive impact, to 5 that have no or only a slight positive impact on the indicator).

Which action will improve the animal's condition and value the most (1) towards least improvement (5)?

MANAGEMENT ACTION	RANK (1 to 5)
ROTATIONAL GRAZING	
REVEGETATION	
CHEMICAL BUSH CONTROL	
NO ROTATIONAL GRAZING	
BUSH ENCROACHED (NO BUSH CONTROL)	

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Linking farmer knowledge and biophysical data to evaluate actions for land degradation mitigation in savanna rangelands of the Molopo, South Africa

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Abstract. The over-utilization of semi-arid savanna rangelands in the North-West Province of South Africa has resulted in profound habitat transformations. A common regional indicator of rangeland deterioration is the imbalance in the grass:woody ratio characterized by a loss of grass cover with increased shrub or tree density. This can result in profound reductions of rangeland productivity forcing farmers to apply active or passive actions to improve rangeland condition to mitigate economic losses. This study forms part of the multinational EU-project PRACTICE (Prevention and Restoration Actions to Combat Desertification: An Integrated Assessment) and aims to evaluate locally applied restoration and management actions using a participatory approach. Actions included rotational grazing, chemical control of woody species and re-vegetation with grasses, and were evaluated by common and site-specific indicators suggested by the farming community. Members of an identified multi-stakeholder platform ranked these indicators according to their relative importance, and results were combined with biophysical measurements for each indicator in a multi-criteria decision analysis. Preliminary results showed rotational grazing management and re-vegetation actions perform equally well in maintaining and restoring an open savanna with a high forage production, followed by selective shrub control. This type of participatory assessment helps to identify best practices, but there is still an urgent need to create legal policy frameworks and institution-building to support local-level implementation in all socio-ecological and economic settings, particularly in communal areas.

Keywords: Best practice, stakeholder participation, indicator identification, shrub encroachment, Kalahari.

Introduction

Approximately 65% of South Africa's rangelands are situated within arid and semi-arid regions and are subjected to infrequent rainfall events, resulting in unpredictable fluctuations in plant production (Snyman 1998). The over-utilization of these rangelands for extended periods can decrease ecosystem resilience and may result in profound habitat transformations (Ibáñez *et al*. 2007). Savanna ecosystems are particularly threatened by a temporary or permanent imbalance in the grass:woody ratio in response to mismanagement (*e.g.* Kgosikoma *et al.* 2012). The underlying process of shrub encroachment and an associated replacement of palatable with unpalatable grasses results in a decrease of biodiversity, rangeland productivity and carrying capacity (Richter *et al.* 2001; Smet and Ward 2005). This has significant socio-ecological implications for land users and forces them to apply active or passive actions to improve rangeland condition and compensate for loss of economic value.

There is a need in South Africa for an information base assisting land users in sustainable land management (Von Maltitz 2009). This can be best achieved through an integrated approach that combines local knowledge with scientific expertise and actively involves land users in evaluation, decision-making and execution processes (Fraser *et al.* 2006; Reed *et al*. 2006). The multinational EU-funded project PRACTICE (Prevention and Restoration Actions to Combat Desertification: An Integrated Approach; www.ceam.es/practice) responded to this general gap and suggested a bottom-up approach based on a participatory and integrated evaluation of local-level land management strategies and restoration actions to combat rangeland degradation (Rojo *et al*. 2012). A multi-step participatory protocol was developed and tested in selected dryland sites worldwide to promote social learning through knowledge exchange by integrating local and expert knowledge and assessments that capture biophysical and socio-economic criteria (Bautista and Orr 2011). Here, we report its application in the savanna rangelands of the semi-arid Molopo region in the North-West Province of South Africa, forming part of the southern Kalahari. Presented results

are preliminary and highlight selected aspects of the integrative assessment approach.

Methods

The evaluation of management and restoration actions applied by local farmers in the study area followed the PRACTICE Integrated Assessment Protocol (for details please refer to Bautista and Orr 2011). Semi-structured interviews were used to identify: (1) a multi-stakeholder platform (MSP); (2) management and restoration actions; and (3) site-specific indicators for action evaluation. Indicators were ranked by members of the MSP according to their perceived importance using a pack-of-cards method and weightings computed *sensu* Figueira and Roy (2002). Indicators related to rangeland productivity and biodiversity were quantified based on biophysical data assessments using the Fixed Point Monitoring of Vegetation (FIXMOVE) methodology (Morgenthal and Kellner 2008). Site selection followed a preferential sampling design guided by the local stakeholders (SHs). A multi-criteria decision analysis (MCDA) conducted with ELECTRE IS (Aït Younes *et al.* 2000) was applied to integrate ranking results and biophysical data for pairwise comparisons of action performances. Reported statistics were carried out using PAST (Hammer *et al.* 2001).

Results and Discussion

The identified MSP consisted of 45 local SHs with different professional backgrounds (Table 1). The conducted interviews with members of the MSP revealed that the most often applied actions to mitigate land degradation in the study area include: (1) rotational grazing management (RGM); (2) chemical shrub control (CSC); and (3) re-vegetation with indigenous grass species (RV).

A short-listing of environmental and socio-economic indicators proposed by the interviewees and a selection of expert-based indicators resulted in a condensed list of 11 indicators for action evaluation (Fig. 1a). The computation of the indicator prioritization process showed that the indicators forage production, grazing capacity and income and profit were ranked highest. Interestingly, local land users perceived the abundance of woody species a less important indicator for evaluating management and restoration impacts (rank 9, Fig. 1a), although there was a clear negative relationship between woody density and grass phytomass as the main contributor to overall forage production (Fig. 1b). This is surprising as degradation indicators related to the density of certain shrub or tree species are commonly used in other parts of the Kalahari (Reed *et al.* 2008). Risks, such as fire or re-vegetation failure, were ranked as least important.

The quantitative assessments revealed that highest tree densities (converted into tree equivalents (TE) *sensu* Teague *et al.* 1981) were found under poor rangeland management (PM; here used as a benchmark), which largely refers to overstocking and no resting periods for vegetation. Accordingly, forage production in poor managed systems was significantly reduced (Table 2). CSC was shown to be important in the transformation of rangelands back into a condition similar to that under RGM with respect to woody density and forage

Table 1. Composition of the multi-stakeholder platform identified in a local consultation process.

Type of expertise	Farmer	Governmental expert	Service provider	Academic	Conservation
Stakeholder category	commercial-private (9) semi-comm.-lease (4) small scale-communal (12) small scale-LRAD* (6)	extension officer (5) researcher (5)	consultant (2)	researcher (1)	manager (1)

*LRAD = Land Redistribution for Agriculture Development

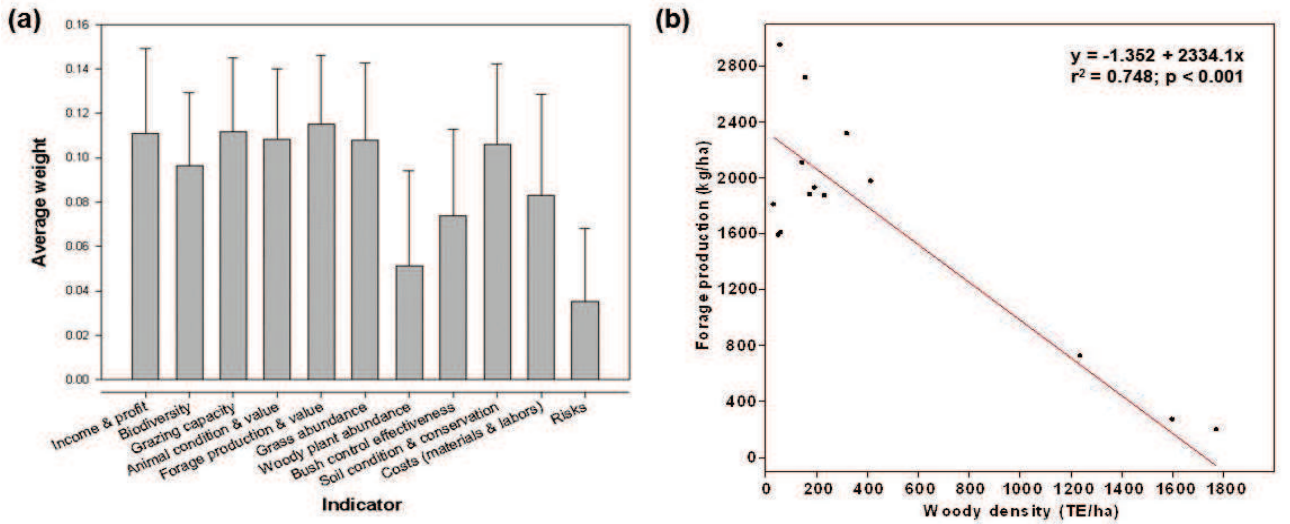

Figure 1. (a) Relative importance of identified indicators averaged over individual stakeholder perceptions, and (b) relationship between the two indicators woody density and forage production (linear model 2: reduced major axis regression).

Table 2. Effect of management and restoration actions as compared to poor management on selected parameters related to identified indicators used for action evaluation. Means (±SD) with different letters in a row indicate a significant difference at *P*<0.05 (ANOVA with post-hoc Tukey's HSD test).

	Rotational grazing (RGM)	Chemical control (CSC)	Re-vegetation (RV)	Poor management (PM)
Woody density (TE/ha)	260.6 ± 87.1 a	252.2 ± 116.3 a	44.4 ± 13.8 b	1531.8 ± 322.6 c
Woody species richness	5.8 ± 1.7 a	6.2 ± 1.7 a	3.3 ± 1.5 ab	8 ± 0 ac
Forage production (kg/ha)	2203.9 ± 328.5 a	1866.6 ± 249.8 a	2120.1 ± 730.1 a	370.7 ± 241.6 c
Grass species richness	6 ± 1.8 a	6.3 ± 4.2 a	5.7 ± 2.9 a	4.3 ± 2.1 a

production. Lowest woody densities were found where the rangeland was re-vegetated, which can be explained by the associated complete clearance of all woody plants. Grass species richness was not significantly affected by management and restoration actions but PM resulted in the lowest grass species richness (Table 2).

The MCDA based on the relevancy (local perception) and performance (biophysical assessment) of actions revealed that in pairwise comparisons RV outranks both CSC and PM, but is as equally good as RGM. The determining criteria were obvious as both these actions (RGM and RV) had the highest measured forage production, which in addition was the first ranked indicator averaged over the MSP. Forage production is also directly related to other indicators perceived as very important, such as income and profit, grazing capacity and animal condition. However, it is clear that to apply a sustainable land management strategy such as RGM, the rangeland has to be open, *i.e.* shrub encroached vegetation states first have to be thinned out. Apart from financial constraints, the choice of the control technology then also depends on the specific land-use objective. RV with its complete clearance of trees and shrubs is an extreme management intervention eliminating any competitive effects in favor of an increased phytomass production of grasses, and thus may be profitable particularly for commercial cattle ranchers. This management may also create open spaces needed on hunting farms, which in addition to having aesthetic value, play an important role in the tourism sector. On the other hand, the selective chemical control of certain increaser shrubs and trees may provide a more balanced approach, and retain important key resources for browsing herbivores such as goats or game.

Conclusions

Although the PRACTICE approach still has to be tested with a complete data set for the Molopo study area, these preliminary results indicate this type of participatory assessment may help to identify best practices. The stakeholder's perspective and circumstances may have a direct influence on the outcomes and contributes to the overall acceptance of results among land users. However, this aspect is likely to be impacted by a social learning effect, which will be verified during an upcoming workshop with members of the MSP aiming at the re-evaluation of actions following group discussions of the preliminary results. The technical implementation of actions will depend on the land-tenure types and management objectives under consideration. While the tested approach is certainly of direct benefit for farm owners, in communal farming systems both a sustainable rangeland management and shrub control are hard to implement. This is due to inappropriate governance structures, strong competition over resources and the high associated costs for materials such as fences and chemicals, respectively. This highlights the urgent need to create legal policy frameworks and institution-building supporting the local-level implementation in all socio-ecological and economic settings.

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